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Nutrient Reconstructions in Standing Waters: Final Report.

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A Report to English Nature by Ensis Ltd. on

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Executive Summary

1. This is the final report to English Nature by ENSIS Ltd under contract F80-11-02: Nutrient Reconstructions in Standing Waters.

2. This project aims to employ palaeolimnological methods to determine a series of lakes which have remained largely unchanged since the 1920/30s, representing a trophic series in a near natural condition. The aim is to use these as model lakes to provide restoration targets for polluted waters.

3. This report describes the current macrophyte flora and water chemistry of thirteen lakes of conservation interest, and the lithostratigraphy, radiometric dating and fossil diatom assemblages in approximately four levels of a sediment core from the deep basin of each lake.

4. The palaeolimnological technique of diatom transfer functions is used to investigate the onset, rate, extent and possible causes of eutrophication at the study lakes. A diatom-based transfer function is applied to the fossil diatom data to generate a quantitative reconstruction of total phosphorus (TP) concentrations for each lake, following taxonomic harmonization between the model training set and the core species data. The TP reconstruction is calculated using a Northwest European training set of 152 lakes (Bennion *et al.*, 1996).

5. In addition, quantitative reconstructions of lake pH were carried out for four sites. These were Wastwater, Oak Mere, Clarepool Moss and Hatchet Pond. pH reconstructions were carried out at these lakes because they were the only sites considered to be sensitive to surface water acidification. The pH reconstructions were calculated using the Surface Water Acidification Programme (SWAP) calibration set of 167 lakes from the UK and Scandinavia (Stevenson *et al.*, 1991). The reconstructions were performed for four approximate points in time: 1850, 1900, 1920/30 and the present day.

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1. Study Rationale and Objectives

Many lakes and reservoirs have exhibited changes in macrophyte communities and a general lowering of biodiversity in recent decades, which suggests that a process of nutrient enrichment is taking place. Phosphorus (P) is one of the key elements which, if in excess, will enrich water and bring about these changes. Another process that has been affecting standing waters, particularly those on base-poor geology in upland regions, is acidification which can also lower biodiversity.

Sewage or animal slurry contains large amounts of P. There is no fixed point in time when enriched standing waters began to show signs of change and, unlike atmospheric pollution, eutrophication is a site specific problem. One event in the history of land use change was the start of the Industrial Revolution in the 1850s. This, with increased centres of urban population and the construction of sewerage systems to transport human waste, certainly led to river pollution, and industrialistion seems to have begun the process of acidification. Additionally farming systems began to change with the advent of tiled drainage and the introduction of artificial fertilisers around the turn of the century. This change could have added to the enrichment effect. By the 1920s and early 1930s farming practices had stabilised but these declined during the Depression years. During the Second World War agriculture was intensified. This process continued until the early 1990s.

Although many standing waters will have been enriched, a few still remain in a semi-pristine state. Similarly, many poorly buffered lakes and tarns in the uplands will have acidified, but a few remain non-acidified.

The Environment Agency (EA) have funded a Lake Classification and Monitoring Study which considers the above impacts upon standing waters in England and Wales. The EA study aims to develop a classification methodology based not solely on the current status of the waters but also incorporates the concept of degree of change, here termed a state-changed scheme. This involves typing a series of lakes using export-coefficients of total P (TP) based on standardised catchments, and the use of several key parameters such as outflow TP, lake depth and retention times (Johnes *et al.*, 1994). The resulting models will be used to hindcast TP concentrations for the 1920/30s by adapting the export-coefficients for low intensity farming as documented in public records of livestock numbers and historical land use maps. These conditions would establish a realistic P baseline for that lake, which could then act as a target value for restoration schemes.

An alternative way to assess the degree of change at a lake is to use palaeolimnological techniques. The lake sediment provides a record of changes that have occurred at the site and diatoms, which preserve well in sediments, are particularly useful indicators of both ecological and chemical change. Models, known as transfer functions, have been developed by the Environmental Change Research Centre (ECRC) and others, that allow TP and pH to be reconstructed from fossil diatom assemblages.

The objectives of the current study are three-fold:

1. To employ palaeolimnological methods to determine a series of lakes which have remained largely unchanged since the 1920/30s. These will represent a trophic series in a near natural condition. The aim is to then use these as model lakes to provide restoration targets for polluted waters.

2. The second element is to reconstruct the TP histories of a series of polluted lakes to assess the timing, extent and possible causes of enrichment at each site.

3. Finally, the results of this work will be incorporated into the EA Lake Classification state-changed scheme in early 1998. The reconstructed TP values using the diatom model will be compared with those hindcast by the export-coefficient approach for a selection of sites.

The project will apply the appropriate diatom transfer functions, developed at the ECRC, to fossil assemblages preserved in lake sediment cores to reconstruct TP and, where appropriate, pH values for four approximate points in time: 1850, 1900, 1920/30 and the present day.

In order to achieve a more complete picture of the current status of the lakes, macrophyte surveys will be carried out at each site. Where possible, the current floras will be compared with historical macrophyte surveys to assess the degree of change.

2. Site Selection and Report Structure

A series of standing waters were selected by English Nature in consultation with the ECRC which were thought to represent standard lake types within a near-natural trophic series, falling into two geographical areas of the country, the north and west (N/W), and the south and east (S/E). Site selection criteria largely followed Ratcliffe (1977) so that the sites should include semi-pristine, dystrophic, oligotrophic, mesotrophic, eutrophic, marl and brackish systems, and where possible each lake type should be represented in the two geographical zones. In addition, three sites thought to represent polluted waters were selected. A total of 13 lakes was chosen and they are listed in Table 1.

Table 1 The selected study sites with lake type and grid references

Lake Type	*Zone*	*Grid Reference*	*Site Name*	*County*
Dystrophic	N/W	SJ 435 343	Clarepool Moss	Shropshire
Oligotrophic	N/W	NY 165 060	Wastwater	Cumbria
Oligotrophic	N/W	SJ 574 677	Oak Mere	Cheshire
Mesotrophic	N/W	NY 214 296	Bassenthwaite Lake	Cumbria
Mesotrophic	N/W	NY 774 698	Greenlee Lough	Northumbria
Mesotrophic	S/E	SU 367 016	Hatchet Pond	Hampshire
Eutrophic	N/W	SD 918 874	Semerwater	North Yorkshire
Eutrophic	N/W	SJ 509 078	Betton Pool	Shropshire
Eutrophic	S/E	TG 388 134	Upton Broad	Norfolk
Brackish	S/E	TG 458 201	Martham South Broad	Norfolk
Polluted	N/W	SD 358 969	Esthwaite Water	Cumbria
Polluted	N/W	SJ 430 305	Crose Mere	Shropshire
Polluted	N/W	SD 894 668	Malham Tarn	North Yorkshire

Each site report begins with a summary page showing the TP and, where appropriate, the pH reconstruction graph and a list of the key findings for that lake. This is followed by a brief site description and a table summarising the main chemical characteristics of the lake: pH, conductivity, alkalinity, calcium, total nitrogen and total phosphorus. In most cases, the water chemistry data are from a survey undertaken in 1995-1996 by the Environment Agency as part of the Lake Classification and Monitoring Phase 2 project. The survey involved monthly sampling but only averages of the data are presented here. However, the data for Oak Mere are from a survey carried out by Moss *et al.* (1992, 1994) in 1991-1992 and the data for Hatchet Pond are from a survey of Hatchet Stream by NRA Southern Region over the period 1979-1983. There are no current data available for Clarepool Moss.

The results of the macrophyte survey are presented next. This is followed by the results of the sediment core analyses in the order: lithostratigraphy, radiometric dating, diatom stratigraphy and the environmental reconstructions. Each site report includes a discussion of the data specific to that lake.

The overall, general findings of the project within the context of lake classification and conservation literature are presented in a summary discussion.

3. Methods

3.1 Macrophyte Surveys

The main aims of the aquatic macrophyte survey were:

i) to map the main stands of emergent, floating leaved and submerged vegetation for each lake

ii) to compile as complete a species list as possible for each site, given the time constraint of one day per site at most lakes and two days per site at Esthwaite Water and Bassenthwaite Lake.

A standard methodology using techniques described below, was applied to all of the lakes, although slight adaptations were necessary on a site by site basis depending on the size and bathymetric characteristics of each water body. At all sites, the perimeter was carefully inspected during a walk around the shoreline, or as in the case of Martham South Broad and Upton Broad where this was not possible, from an inflatable boat. At sites where walking was possible, the surveyor regularly waded out to thigh-depth and employed an underwater viewer or bathyscope to detect vegetation.

An inflatable boat was used during the shoreline inspection to investigate deeper water locations and detect plants growing toward their photic limit, and a double headed rake grapnel was used frequently to trawl in a direction perpendicular to the shoreline.

Additional information on the presence of species was provided by the inspection of strand-line material on down-wind shorelines.

At some sites, where zonation of species with depth was evident, fixed transect lines were layed perpendicular to the shoreline and approximately 60 m in length. A combination of Ekman grab, double-headed rake grapnel and bathyscope were used to detect the presence and retrieve samples of the underlying vegetation, and diagrams of zonation were compiled.

Specimens of the more difficult to identify plant groups were collected and preserved according to standard techniques. Charophytes and *Potamogeton* specimens were verified by experts at the British Museum and the Biological Records Office respectively.

For each site, a list was compiled of all aquatic taxa, under the categories of emergent, floatng leaved and submerged. The abundance of each taxon was scored according to the DAFOR scale as used in NCC lake surveys; dominant (D), abundant (A), frequent (F), occasional (O), and rare (R). The aquatic macrophyte classification system of Palmer (1992) was applied to each species list.

3.2 Coring and Lithostratigraphic Analyses

A single sediment core was taken from the deepest basin of each lake during 1995, using either a Glew corer or a Mackereth corer. The cores were extruded in the laboratory at 1 cm intervals and the Troels-Smith sediment description system (Troels-Smith, 1955) was used to describe the main characteristics of the sediment and any stratigraphic changes were noted.

The percentage dry weight (%dw) for each sample was calculated by weighing approximately 1g of wet sediment in a pre-weighed crucible, from each pre-homogenised sediment layer, drying the sediment at 105°C for at least 16 hours, then reweighing the crucible. Approximate organic matter content was then determined (as a percentage loss on ignition - %loi) by placing the crucible containing the dried sediment in a muffle furnace at 550°C for two hours and then reweighing.

The wet density of the sediment is determined by its composition. Variations in density down a core indicate fluctuations in sediment composition suggesting more than one sediment source. Sediment density is also required for calculations of sediment accumulation rate if the core is to be dated. It is measured by weighing an empty 2cm^3 capacity brass phial to 4 decimal places and then filling it with wet sediment. The phial is then re-weighed and the weight of the sediment divided by 2 to determine the density as grams per cm^3.

3.3 Radiometric Dating

^{210}Pb occurs naturally in lake sediments as one of the radioisotopes in the ^{238}U decay series. It has a half-life of 22.26 years, making it suitable for dating sediments laid down over the past 100-150 years. The total ^{210}Pb activity in sediments comprises supported and unsupported ^{210}Pb (Oldfield & Appleby, 1984). In most samples the supported ^{210}Pb can be assumed to be in radioactive equilibrium with ^{226}Ra and the unsupported activity at any level of a core is obtained by subtracting the ^{226}Ra activity from the total ^{210}Pb.

^{210}Pb dates for sediment cores can be calculated using both the constant rate of ^{210}Pb supply (CRS) model and the constant initial ^{210}Pb concentration (CIC) model (Appleby & Oldfield, 1978). The CRS model is most widely accepted; it assumes that the ^{210}Pb supply is dominated by direct atmospheric fallout, resulting in a constant rate of supply of ^{210}Pb from the lake waters to the sediments irrespective of net dry mass accumulation rate changes. If there are interruptions to the ^{210}Pb supply, for example sediment focusing, dates are calculated either by the CIC model or by using a composite of both models. The factors controlling the choice of model are described in full in Appleby & Oldfield (1983), and Oldfield & Appleby (1984).

^{137}Cs activity in sediments prior to the 1986 Chernobyl nuclear accident derives mainly from nuclear weapons testing fallout. Where this isotope is strongly adsorbed on to sediments, the activity versus depth profile is presumed to reflect varying fallout rate and useful chronological markers are provided by the onset of ^{137}Cs fallout in 1954, and peak fallout in 1963.

Sediment sub-samples from selected levels of each core were analysed for ^{210}Pb, ^{226}Ra, ^{137}Cs and ^{241}Am by direct gamma assay in the Liverpool University Environmental Radioactivity

Laboratory, using Ortec HPGe GWL series well-type coaxial low background intrinsic germanium detectors (Appleby *et al.* 1986). ^{210}Pb was determined via its gamma emissions at 46.5keV, and ^{226}Ra by the 295keV and 352keV γ-rays emitted by its daughter isotope ^{214}Pb following 3 weeks storage in sealed containers to allow radioactive equilibration. ^{137}Cs and ^{241}Am were measured by their emissions at 662keV and 59.5keV. The absolute efficiencies of the detectors were determined using calibrated sources and sediment samples of known activity. Corrections were made for the effect of self absorption of low energy γ-rays within the sample (Appleby *et al.* 1992).

^{210}Pb chronologies were calculated using the CRS and CIC dating models (Appleby & Oldfield, 1978). Discrepancies between the two models, where they occured, were resolved where possible using the 1986 and 1963 dates determined by ^{137}Cs and ^{241}Am (Appleby *et al.* 1991) records from Chernobyl and atmospheric nuclear weapons test fallout.

3.4 Diatom Analysis and Transfer Functions

In the absence of long-term historical water chemistry data, the sediment accumulated in lakes can provide a record of past events and past chemical conditions (e.g. Smol, 1992). Diatoms (unicellular, siliceous algae) are particularly good indicators of past limnological conditions, for example lake pH, nutrient concentrations and salinity. In recent years, quantitative approaches have been developed, of which the techniques of weighted averaging (WA) regression and calibration, developed by ter Braak (e.g. ter Braak & van Dam, 1989), are currently the most statistically robust and ecologically appropriate. WA has become a standard technique in palaeolimnology for reconstructing past environmental variables. The methodology and the advantages of WA over other methods of regression and calibration are well documented (e.g. ter Braak & van Dam, 1989; ter Braak & Juggins, 1993; Line *et al.*, 1994).

Using the technique of WA, a predictive equation known as a transfer function can be generated that enables the inference of a selected environmental variable from fossil diatom assemblages, based on the relationship between modern surface-sediment diatom assemblages and contemporary environmental data for a large training (or calibration) set of lakes. This approach has been successfully employed in recent years to quantitatively infer lake pH (e.g. Birks *et al.*, 1990) and lake total phosphorus (TP) concentrations (e.g. Anderson *et al.*, 1993; Bennion, 1994; Bennion *et al.*, 1996), whereby modern diatom pH and TP optima are calculated for each taxon based on their distribution in the training set, and then past pH and TP concentrations are derived from the weighted average of the optima of all diatoms present in a given fossil sample. These models are able to provide estimates of baseline pH and TP concentrations in lakes, and coupled with dating of sediment cores (radiometric or spherical carbonaceous particles), enable the timing, rates and possible causes of acidification and enrichment to be assessed for a particular site. This information can be used to design lake classification systems and can be incorporated into lake management and conservation programmes.

In this study, approximately four levels of a core from each lake were prepared and analysed for diatoms using standard techniques (Battarbee, 1986). At least 300 valves were counted from each sample using a Leitz research quality microscope with a 100 x oil immersion objective and phase contrast. The data were expressed as percentage relative abundance. All slides are archived at the ECRC.

The appropriate transfer functions were applied to the core data to generate quantitative reconstructions of TP for all sites and in some cases also pH, following taxonomic harmonization between the training sets and core species data. pH reconstructions were only carried out at Wastwater, Clarepool Moss, Oak Mere and Hatchet Pond as these are the only sites that are classified as sensitive to acidification based on the geology and soil types (Edmunds & Kinniburgh, 1986; Hornung *et al.*, 1995).

The TP reconstructions were calculated using a Northwest European calibration set of 152 lakes (Bennion *et al.*, 1996; see Appendix), based on simple WA with inverse deshrinking on log-transformed annual mean TP data. The relationship between diatom-inferred (DI-TP) and measured TP is strong ($r^2 = 0.85$) and has low errors of prediction with an apparent root mean square error (RMSE) of 0.19 and a cross-validated RMSE (RMSE-P) of 0.22 (log values), indicating that the model performs well. The pH reconstructions were calculated using the Surface Water Acidification Programme (SWAP) calibration set of 167 lakes from the UK and Scandinavia (Stevenson *et al.*, 1991), using simple WA with classical regression. The pH transfer function is robust with an RMSE-P of 0.32 pH units. All reconstructions were implemented using CALIBRATE (Juggins & ter Braak, 1993).

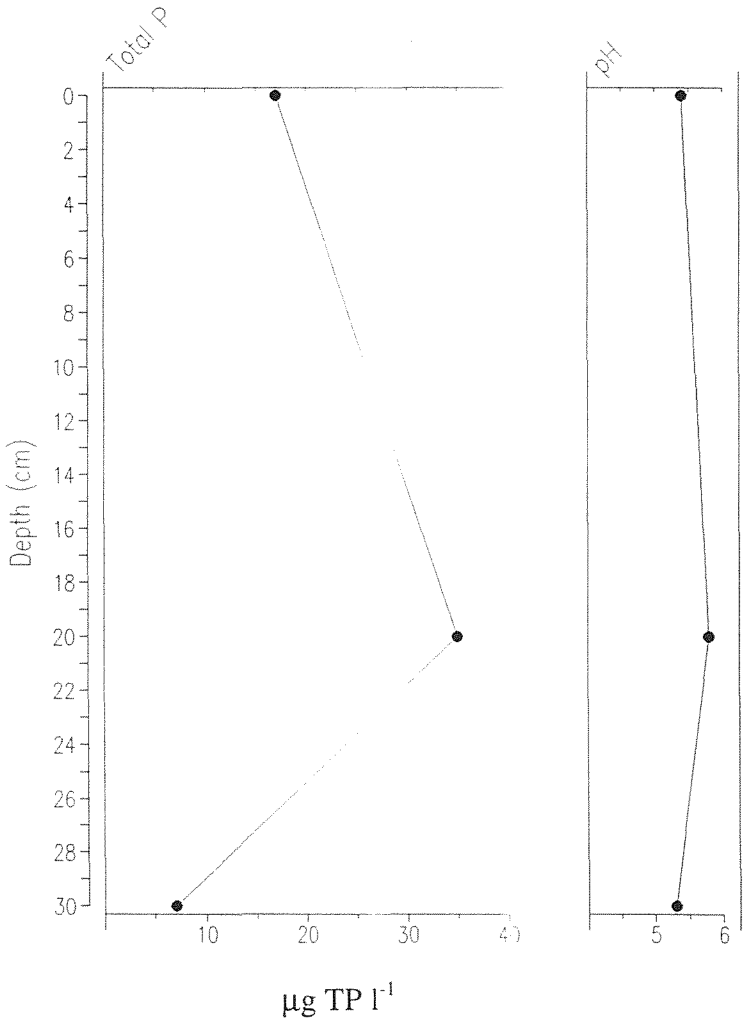
In addition to the quantitative reconstructions, the study also describes the diatom data qualitatively. Diatoms have been recognised as good indicators of lake trophic status and pH for a long time and therefore an interpretation of species shifts in sediment cores can also provide important information on ecological and chemical changes in the lake. Certain taxa are commonly associated with strongly eutrophic waters, such as the small, centric planktonic taxa of the genera *Stephanodiscus* and *Cyclostephanos* (eg. Anderson, 1990; Bennion, 1995), whilst the small *Cyclotella* taxa such as *C. comensis* and *C. glomerata* are typically found in more oligotrophic waters (eg. Wunsam & Schmidt, 1995; Anderson, 1997). Some taxa appear to prefer mesotrophic conditions, such as *Tabellaria flocculosa*, *Fragilaria crotonensis* and a number of the *Aulacoseira* species. There have been numerous palaeolimnological studies from lakes in Europe and the United States which show the gradual replacement of the small *Cyclotellas* by *Aulacoseira* spp., *Tabellaria flocculosa* or *Fragilaria crotonensis*, which in turn are replaced by the small *Stephanodiscus* taxa as the process of nutrient enrichment takes place (eg. Bradbury, 1975; Battarbee, 1978, 1986; Bennion, 1994; Anderson, 1997).

Diatoms also exhibit a response to pH with certain taxa such as *Brachysira brebissonii*, many of the *Eunotia* spp. and *Frustulia rhomboides* being good indicators of very acid conditions, whilst *Achnanthes minutissima* and *Cyclotella kuetzingiana*, for example are more commonly observed in circumneutral waters (Stevenson *et al.*, 1991). Likewise, some diatom taxa are indicative of more brackish conditions, such as *Mastogloia smithii* and *Amphora coffeaeformis* and may signal saline intrusions into freshwater systems.

Species shifts may occur in response to factors other than changes in water chemistry, for example physical disturbance such as soil inwash or water level manipulation, or a loss of habitats, which may be related to water quality changes. A switch from a non-planktonic dominated diatom assemblage with many epiphytic taxa (e.g. *Achnanthes minutissima* and *Cocconeis* spp.) and benthic taxa (e.g. *Fragilaria* spp.) to one dominated by a planktonic community has been observed at many lakes in the UK in response to a loss of the submerged aquatic macrophyte community and reduced light conditions. The most famous examples of these are from the Norfolk Broads (e.g. Osborne & Moss, 1977; Moss, 1980; Bennion, 1996).

Clarepool Moss

- Owing to a lack of similar dystrophic sites in the calibration sets and poor diatom preservation, the diatom models cannot be applied to Clarepool Moss with any confidence.
- The diatom flora, however, does suggest that the pool has always been acidic with dominance of low-pH tolerant species throughout the core.
- There is no clear evidence of nutrient enrichment.



4. Clarepool Moss, Shropshire (SJ 435 343)

4.1 Site Description and Water Chemistry

- Very small area of open water, approximately 100 m by 80 m
- Altitude 90 m
- A shallow (maximum depth 2.2 m) pool lying within one of the Shropshire moss complexes
- A dystrophic site with peat-stained water
- The pool is surrounded by *Juncus effusus*, *Carex panicea* and large areas of *Sphagnum* moss (Schwingmoor).
- The catchment is mostly birch woodland with some pine, alder and willow carr, and arable.

There are no current water chemistry data for Clarepool Moss. The only data that could be found were those from a spot sample survey by Eville Gorham in June 1955 and are thus rather old and should be viewed with caution:

pH	units	5.4
total ions	meq l^{-1}	1.80
calcium	meq l^{-1}	1.23
nitrate	mg l^{-1}	0.05
orthophosphate	$\mu g\ l^{-1}$	80

4.2 Macrophyte Survey

Date of visit: 18-9-95.
Clarepool Moss does not support any floating or submerged vegetation. The pool is surrounded by *Juncus effusus* and *Carex paniculata* with *Molinia* in places and some *Sphagnum.*

4.3 Lithostratigraphy

A 100 cm sediment core (CLAP1) was taken in a water depth of 2.1 m using a piston corer on 18-9-95. The sediment was a homogenous very dark brown (10YR 2/2) unconsolidated, highly humified, organic peat for the whole core length (Ld4 Lso++).

The %dw, %loi and wd profiles (Figure 1) show very little change throughout the whole period represented by the core. The %loi values were extremely high (80-90%) and consequently the %dw and wd values very low (5-10%) because of the organic, peaty nature of the material.

4.4 Radiometric Dating

Very low unsupported ^{210}Pb activities in this core recorded only in the top 5 cm sediments do not allow dating by the ^{210}Pb method. The ^{210}Pb inventory is estimated to be c.100 Bq kg^{-1}, less than 10% of the fallout record.

Moderately high ^{137}Cs activities were recorded down to 20 cm depth, though there is no indication of the level of the 1963 weapons fallout peak.

On the basis of these results, illustrated in Figure 2 and Figure 3, it is not possible to make any realistic estimate of recent accumulation rates.

4.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in three levels of the sediment core were calculated and Figure 4 illustrates the results for the major taxa. Diatom preservation was poor, except in the surface sediment sample, with progressively poor preservation downcore. Analysis was not possible below 30 cms. A total of 52 taxa was observed, 35 of which were present in the TP calibration set. There were analogue problems throughout the core owing to the relative importance of acid-tolerant diatom species in the fossil assemblages that were not present in the TP calibration set. Analogues were particularly poor for the surface sample where only 40% of the data could be used in the TP calibration procedure. There were also analogue problems for the pH reconstruction with only 31 of the fossil taxa present in the pH calibration set, and only 70% and 52% of the fossil data could be used in the pH reconstructions for the surface and 10 cm sample respectively.

Furthermore, diatom dissolution was extremely bad in the 30 cm sample with presence of only very robust forms. The data may not therefore represent the true assemblage for this time and the observed changes in species composition may be more a reflection of the standard of preservation than any real species shifts. In view of these analogue problems, the reconstructions should be interpreted with caution.

Figure 4 illustrates changes in the diatom species composition over the period represented by the upper 30 cm of the core. Radiometric dating results indicate problems in determining sediment accumulation rates and thus no chronology has been obtained. The bottom sample (30 cm) was dominated by *Eunotia incisa* and *Frustulia rhomboides*, taxa commonly observed in acid waters. The 20 cm sample appeared more diverse but this was a function of better preservation. The dominant species was a fine, poorly silicified form of *Aulacoseira granulata,* and those taxa that were present in the lower sample were still important. The surface sample was quite different with high frequencies of *Eunotia meisteri*, *Eunotia curvata*, *Pinnularia appendiculata* and *Pinnularia interrupta,* taxa associated with acid conditions.

4.6 Total Phosphorus and pH Reconstructions

The TP reconstruction shows that there have been changes in the nutrient status of Clarepool Moss. Given the analogue and dissolution problems, however, the diatom-inferred results are not reliable and only a qualitative interpretation of the species changes can be made. The diatom-inferred TP (DI-TP) values produced by the model ranged from 7 $\mu g\ l^{-1}$ for the bottom sample to 35 μg TP l^{-1} for the 20 cm sample, decreasing again to 17 μg TP l^{-1} for the 1995 sample. The lower TP value for the bottom sample is explained by the high abundances of *Eunotia incisa* and *Frustulia rhomboides* which have very low TP optima, but the dominance of these two taxa is likely to be related to the degree of preservation, as discussed above, and the model may therefore be under-estimating the TP concentrations. The higher estimated TP value for the 20 cm sample is associated with the rise in *Aulacoseia granulata*, which can tolerate high nutrient concentrations and thus has a high TP optimum.

The pH reconstruction results must also be viewed in light of the analogue difficulties. The diatom-inferred pH (DI-pH) for the three levels were very similar, ranging from 5.3 for the bottom sample to 5.8 for the 20 cm sample, and pH of 5.4 for the surface sample.

4.7 Discussion

There are no water chemistry records available for Clarepool Moss and therefore it is not possible to validate the results of the reconstructions. A spot water sample taken by Gorham in 1955 gave a pH value of 5.4 and therefore the pH values produced by the model of 5.3-5.8 appear to be in the correct range (F. Rose, pers. comm.). The pool is classed as dystrophic because of the dominance of peat and the water colour is very brown with a secchi depth of only 18 cm. There are no similar dystrophic lakes in the northwest European calibration set and therefore the diatom models cannot be used with any confidence to reconstruct the TP or pH history of Clarepool Moss. The interpretation is also hindered by diatom dissolution and the lack of a chronology for the core. The diatom flora, however, does suggest that the lake has always been acid with dominance of acidophilous (low-pH tolerant) species throughout the core. The diatoms appear to be responding more to the low pH of the water rather than to nutrient concentrations and thus there is no clear evidence of enrichment. The results confirm that Clarepool Moss is a dystrophic system but it is probably not analogous to many other standing waters in the UK.

Figure 1 Lithostratigraphic data for Clarepool Moss

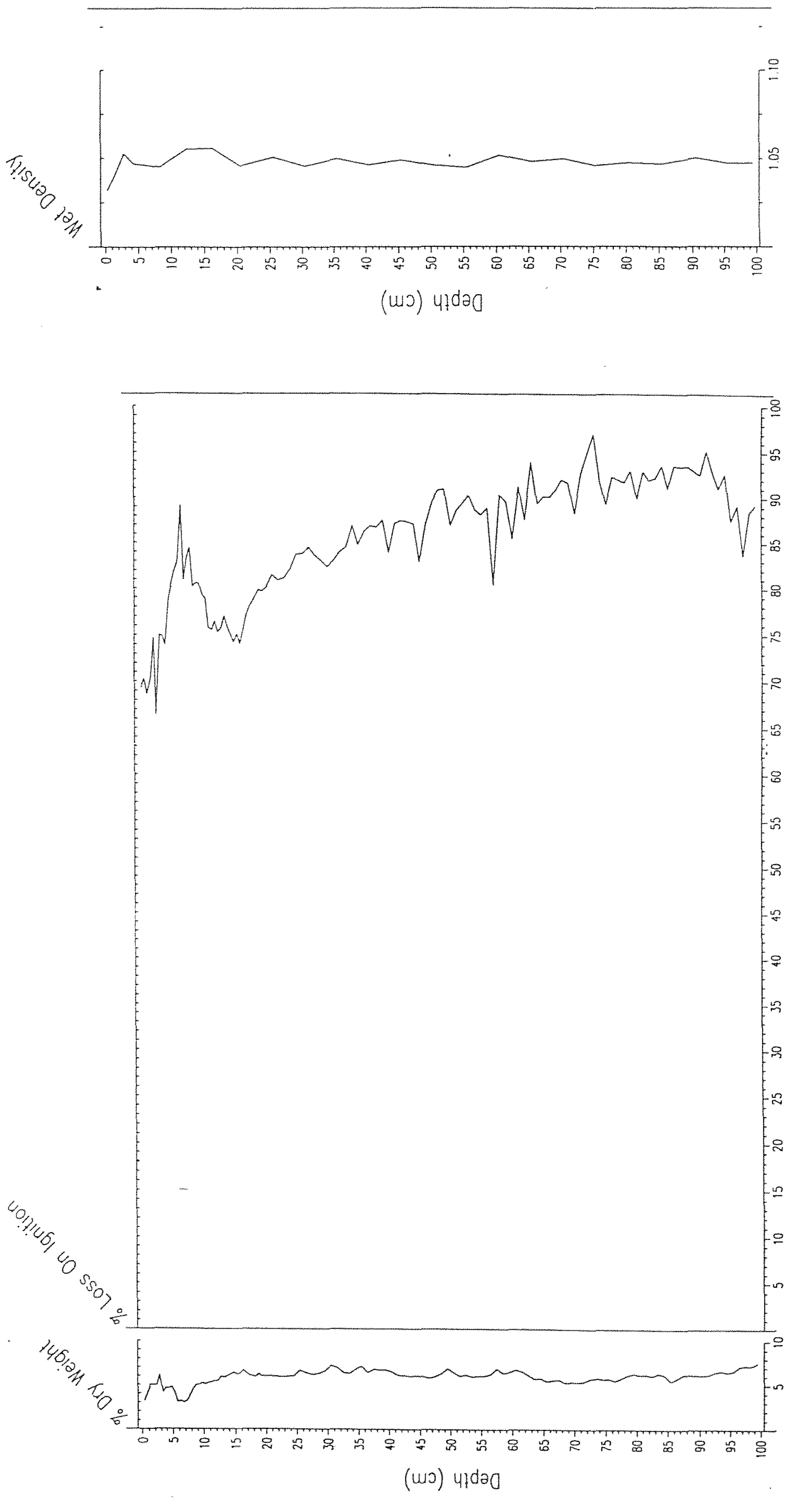


Figure 2 ^{210}Pb Activity versus Depth - Clarepool Moss

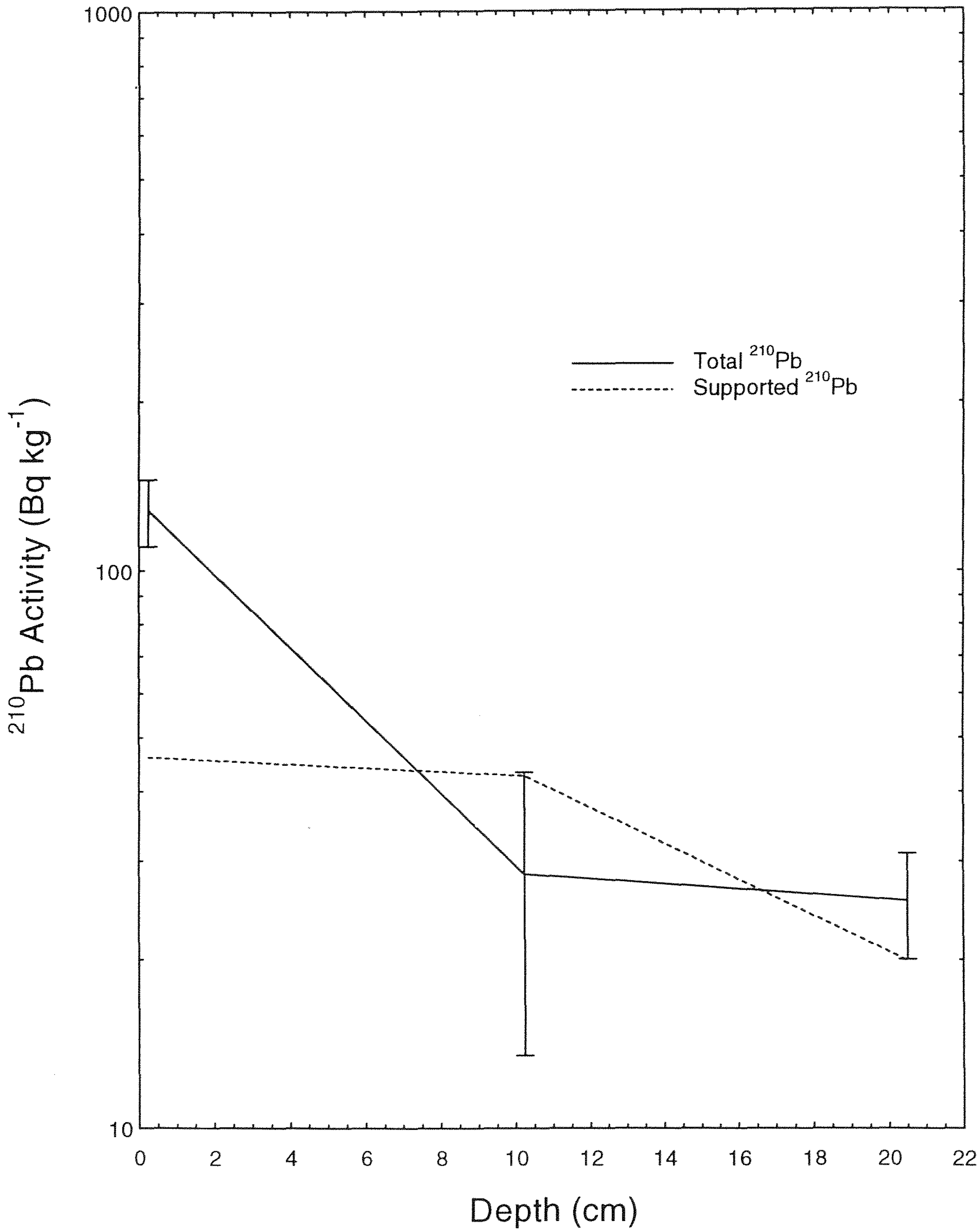


Figure 3 ^{137}Cs Activity versus Depth - Clarepool Moss

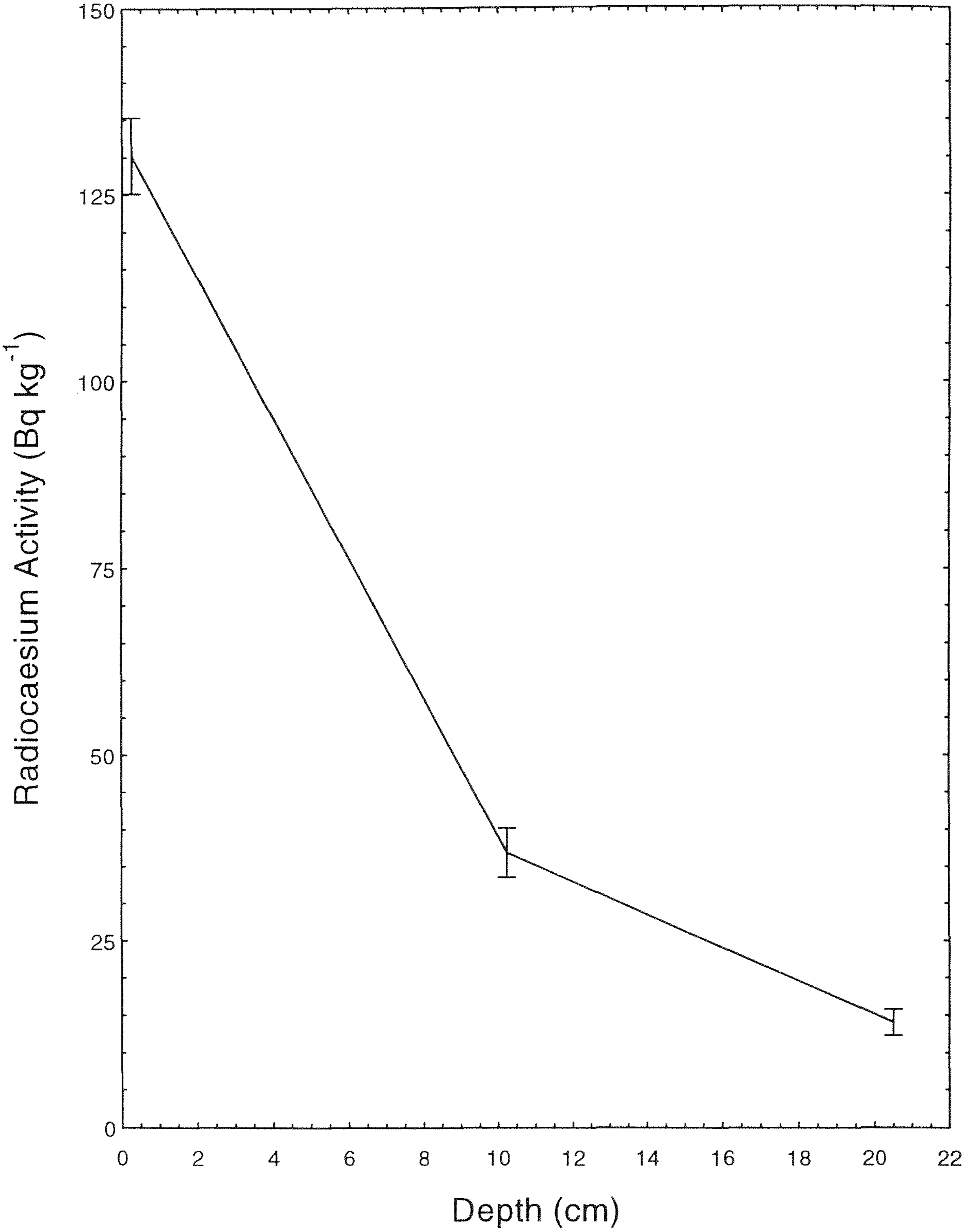
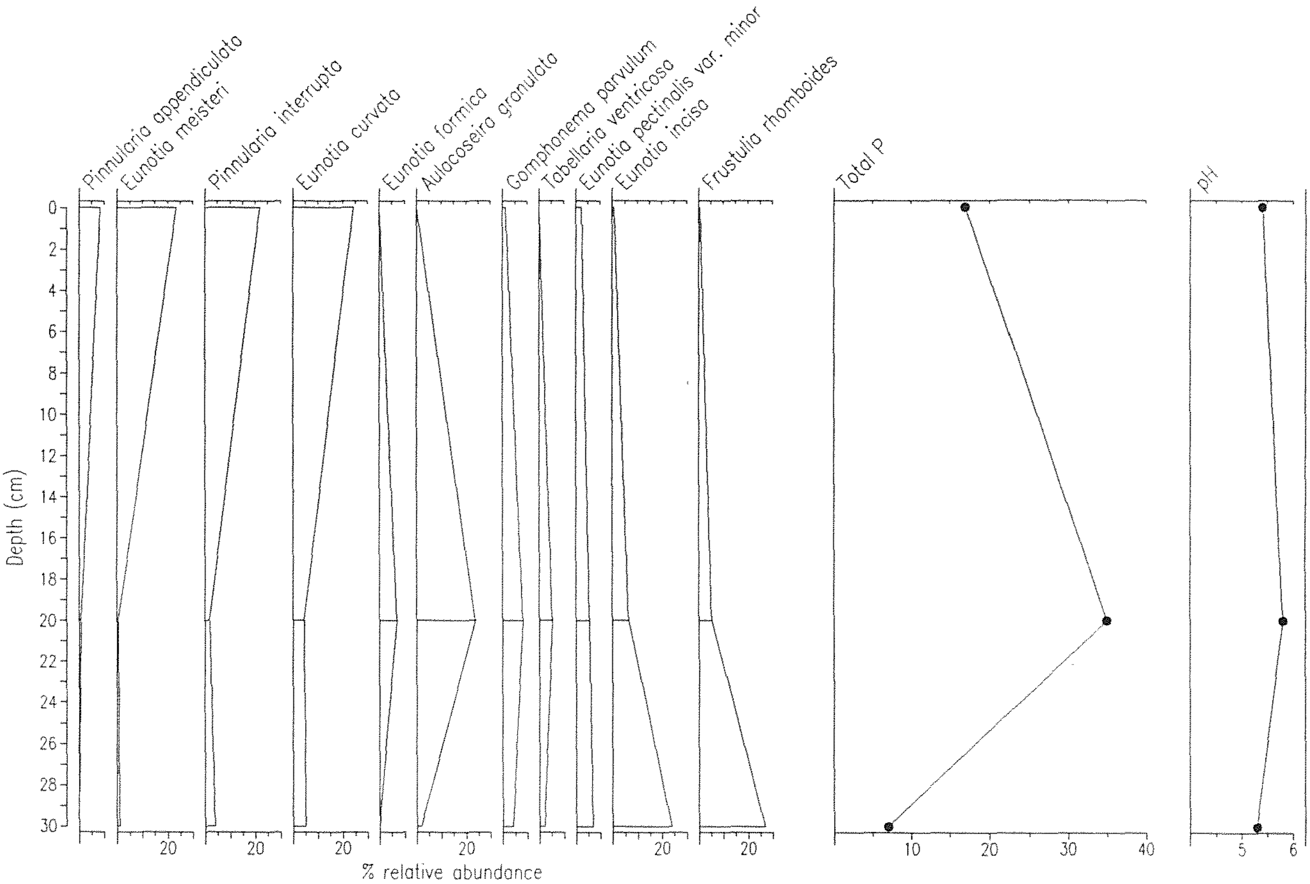
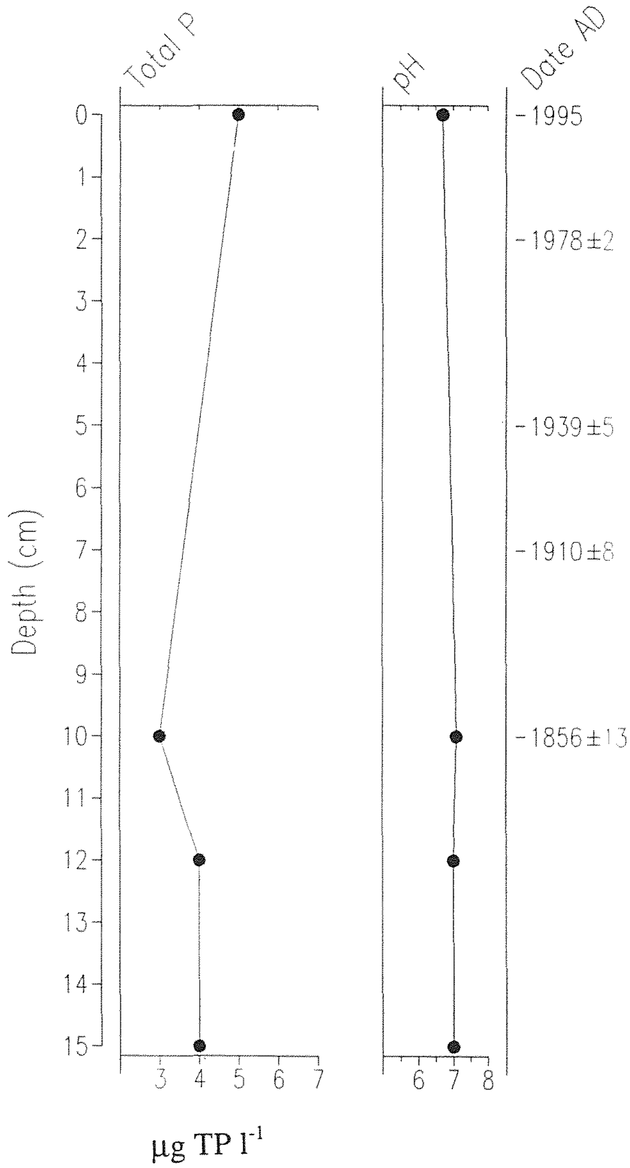


Figure 4 Summary diatom diagram and reconstructions for Clarepool Moss



Wastwater

- Wastwater represents a typical oligotrophic lake in a near natural condition.
- There were no significant changes in either DI-TP or DI- pH.



5. Wastwater, Cumbria (NY 165 060)

5.1 Site Description and Water Chemistry

- Wastwater is the deepest lake in England and the least productive lake in the Cumbrian Lake District
- Altitude 61m
- Area 2.91 km^2
- Mean depth 39.7 m
- Maximum depth 76 m
- Approximately 5% of the total drainage area is cultivable.

pH	units	6.8
conductivity	$\mu S\ cm^{-1}$	46
alkalinity	mg $CaCO_3$	5
calcium	mg l^{-1}	2.4
total nitrogen	mg l^{-1}	0.47
total phosphorus	$\mu g\ l^{-1}$	< 10

nb. These readings are taken from the lake outflow (River Irt)

5.2 Macrophyte Survey

Date of visit 5-7-96.
The aquatic macrophyte flora of Wastwater is typical of oligotrophic lakes in the UK which have not undergone significant acidification. Much of the shoreline is steeply shelving and exposed to wave action and is therefore devoid of organic substrates suitable for the establishment of aquatic plants. However, significant colonisation occurs below approximately 0.5 m water depth on the north-west side of the lake, and, owing to the exceptional water clarity at this site, vascular plants penetrate to a depth of 7.0 m. A map of aquatic macrophyte distribution in Wastwater is presented in Figure 5 and a transect profile is presented in Figure 6. A species list is provided in Table 2.

Although locally dependent on exposure, the same aquatic macrophyte communities can be identified along most of the north-west side of the lake. Shallow water is dominated by an association of *Littorella uniflora* and *Lobelia dortmanna* to a water depth of approximately 2.5 m, below which *Myriophyllum alterniflorum* and *Juncus bulbosus* var. *fluitans* are more abundant. The charophyte *Nitella flexilis* var. *flexilis* which is indicative of only slightly acid conditions, is also locally abundant in this deeper habitat.

Below approximately 3.5 m water depth, *Isoetes lacustris* becomes the dominant species and is the only aquatic vascular plant between 5.0 m and 7.0 m. *I. lacustris* is known for its ability to grow at low light levels as a result of its low compensation point, however it is rarely found at such depths. Preston and Croft (1997) state the maximum depth of *I. lacustris* in the UK to be only 6.0 m as recorded by West (1905) at Loch Lundie.

The relatively sheltered north and south ends of the lake contain a more diverse flora. At the north end *Ranunculus flammula* is common amongst a sward of *L. uniflora* in shallow water. *Fontinalis antipyretica* is also locally abundant and a stand of *Carex rostrata* occupies a small wind protected shallow bay.

The sheltered bay within the neck of the lake outflow also contains an association of *L. uniflora* and *R. flammula*, with *J. bulbosus* var. *fluitans*, *M. alterniflorum* and *I. lacustris* other common components of the assemblage.

The south-west shoreline, beneath the screes of Lligill Head, shelves steeply and is too unstable to allow colonisation of aquatic macrophytes in most locations, although *I. lacustris* does occur in isolated patches in deeper water.

The species list is very similar to that of Stokoe (1983) which covers four surveys of Wastwater between 1975 and 1980. The only notable difference is that there is a record for *Potamogeton obtusifolius* for this period. However, it is quite possible that this species was overlooked during the current survey and there is no evidence to suggest a change in the composition of the macroflora over the past few decades.

Applying the species list in Table 2, Wastwater is classified as Type 3 (Palmer, 1992), an oligotrophic category with predominantly coarse substrates.

Table 2 Species list and DAFOR abundance rating for Wastwater

	abundance	notes
Submerged taxa		
Nitella flexilis var. *flexilis*	O	locally abundant off n-w shoreline
Fontinalis antipyretica	O	locally abundant in shallow water at n end
Callitriche sp.	R	in shallow water at n end
Isoetes lacustris	A	dominant at water depth between 3.5 - 7.0 m
Juncus bulbosus var. *fluitans*	F	locally abundant off n-w shoreline
Littorella uniflora	A	shallow water - except s-e shoreline
Lobelia dortmanna	A	shallow water - except s-e shoreline
Myriophyllum alterniflorum	F	locally abundant off n-w shoreline
Ranunculus flammula	O	locally abundant in shallow water - n and s end
Emergent taxa		
Carex rostrata	R	in shallow bay at n end

5.3 Lithostratigraphy

A 38 cm sediment core (WAST1) was taken from the deepest part of the lake at a water depth of 76 m using a Glew corer on 6-6-95. The lower part of the sediment core was dark brown (10YR 3/3) lake mud with some clay and plant fragments (Ld2 Lso2 As++ Dh+). There was a slight colour change at 10 cm and the upper 10 cms were a very dark brown (10YR 2/2), organic lake mud (Ld3 Lso1 Dg+ Dh+).

The percentage dry weight (%dw) and loss on ignition (%loi) profiles (Figure 7) show that %dw has fluctuated at around 20% and %loi at around 15% for most of the period represented by the sediment core. However there was an increase in percentage organic matter in the upper 10 cm, %loi rising to 25% at the sediment surface. There was very little change in the wd throughout the core (values range from 1.0-1.2 g cm^{-3}).

5.4 Radiometric Dating

The Wastwater core had a relatively simple ^{210}Pb profile with a good record spanning the past 140 years. Equilibrium with the supporting ^{226}Ra was reached at a depth of c.12 cm. There was little significant difference between dates calculated using the CRS and CIC ^{210}Pb dating models. The mean sedimentation rate for the past 140 years was estimated to be 0.011±0.001 g $cm^{-2}y^{-1}$ (0.07 cm y^{-1}).

High ^{137}Cs activities in the top 2 cm record fallout from the 1986 Chernobyl accident, and a well defined ^{241}Am peak at 3.25±0.5 cm records the 1963 fallout maximum from the atmospheric testing of nuclear weapons. Both these dates are in relatively good agreement with those determined by ^{210}Pb.

The results are summarised in Table 3 and Figure 8, Figure 9 and Figure 10.

Table 3 Chronology of Wastwater

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0.0	0.00	1995	0				
1.0	0.07	1988	7	1		0.12	
2.0	0.18	1978	17	2		0.09	
3.0	0.32	1965	30	3		0.08	
4.0	0.45	1953	42	4		0.08	
5.0	0.60	1939	56	5	0.011	0.07	9.3
6.0	0.75	1925	70	7		0.07	
7.0	0.91	1910	85	8		0.06	
8.0	1.09	1893	102	9		0.06	
9.0	1.29	1875	120	11		0.06	
10.0	1.49	1856	139	13		0.05	

5.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in four levels of the sediment core were calculated and Figure 11 illustrates the results for the major taxa. Diatom preservation was good throughout the core. A total of 77 taxa was observed, 53 of which were present in the TP calibration set and 45 in the pH calibration set. Most of the common taxa were well represented in the TP calibration set, with greater than 80% of the fossil assemblage being used in the calibration procedure, except for the 10 cm sample where only 74% of the fossil data was used. This was due to the high relative abundance of an unknown species of *Cyclotella*, referred to as *Cyclotella* unknown sp. 1, which was absent from the calibration set. Similarly, over 78% of the fossil data was used in the pH reconstructions, except for the 10 cm sample.

Figure 11 illustrates that there has been little change in the diatom species composition over the period represented by the upper 15 cm of the core (c. 1850-1995). The small, planktonic *Cyclotella* taxa, especially *C. comensis*, *C. cyclopuncta* and *C.* unknown sp. 1, species generally found in oligotrophic waters, have dominated the assemblages. *Achnanthes minutissima* and *Brachysira vitrea*, non-planktonic species typical of waters low in nutrients, were also frequent throughout the core.

5.6 Total Phosphorus and pH Reconstructions

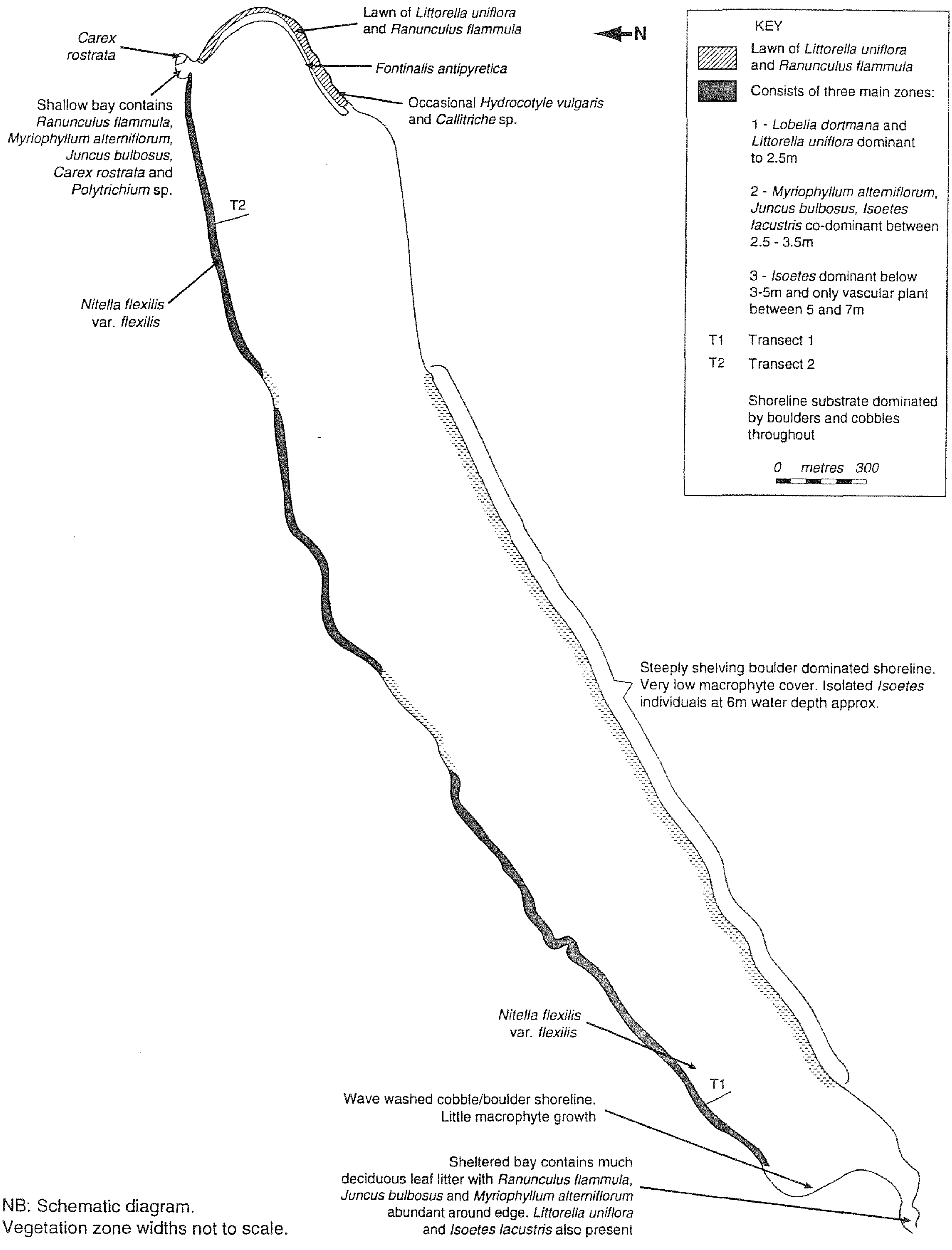
The TP reconstruction shows that the lake has been oligotrophic for the whole of the period from around 1850 to the present day with no marked change in TP concentrations. The model provides an estimate of c. 3-5 μg TP l^{-1} for all four samples.

Similarly, there was no change in diatom-inferred pH over the period 1850-1995 with values of c. pH 6.8-7.0 for all samples.

5.7 Discussion

These findings are consistent with Pearsall's evolutionary series of the English Lake District lakes (Pearsall, 1921), where Wastwater was defined as a Group 1, rocky, unproductive lake with only 5% of cultivable drainage area. Therefore, one would expect the lake to be one of the most pristine waterbodies in the UK. The DI-TP value of c. 5 $\mu g\ l^{-1}$ for the surface sample compares well with the current, measured TP of < 10 $\mu g\ l^{-1}$. Likewise, the DI-pH of 6.8 corresponds exactly with the measured mean pH of 6.8 (annual range 6.2-7.3). The close match between the model results and field data suggests that the reconstructions for Wastwater are reliable. The dating results indicate that mean sediment accumulation rates have remained largely unchanged for the last 150 years or so. The palaeolimnological data, therefore, confirm that Wastwater represents a typical oligotrophic lake in a "near natural" condition.

Figure 5 Aquatic macrophyte distribution map for Wastwater



NB: Schematic diagram.
Vegetation zone widths not to scale.

Figure 6 Aquatic macrophyte transect profile for Wastwater

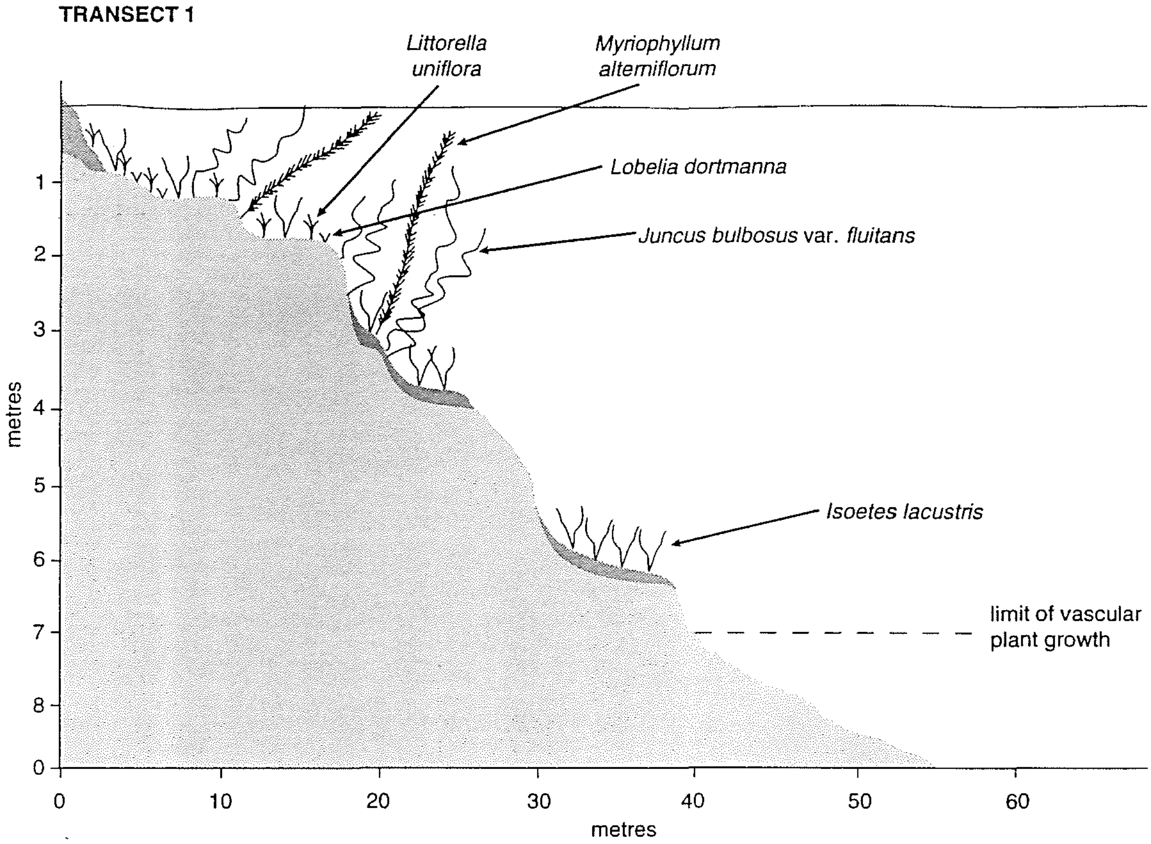


Figure 7 Lithostratigraphic data for Wastwater

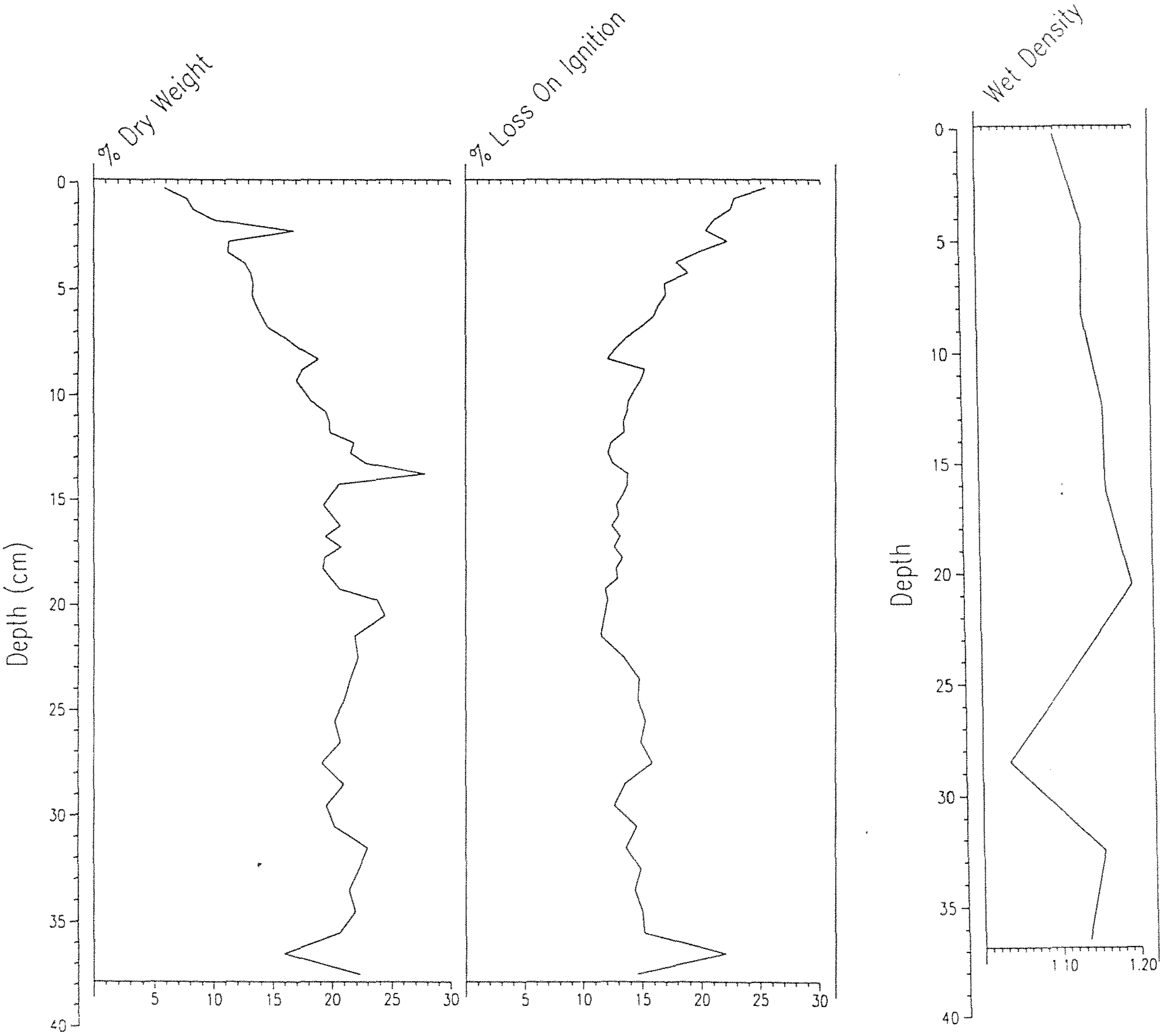


Figure 8 ^{210}Pb Activity versus Depth - Wastwater

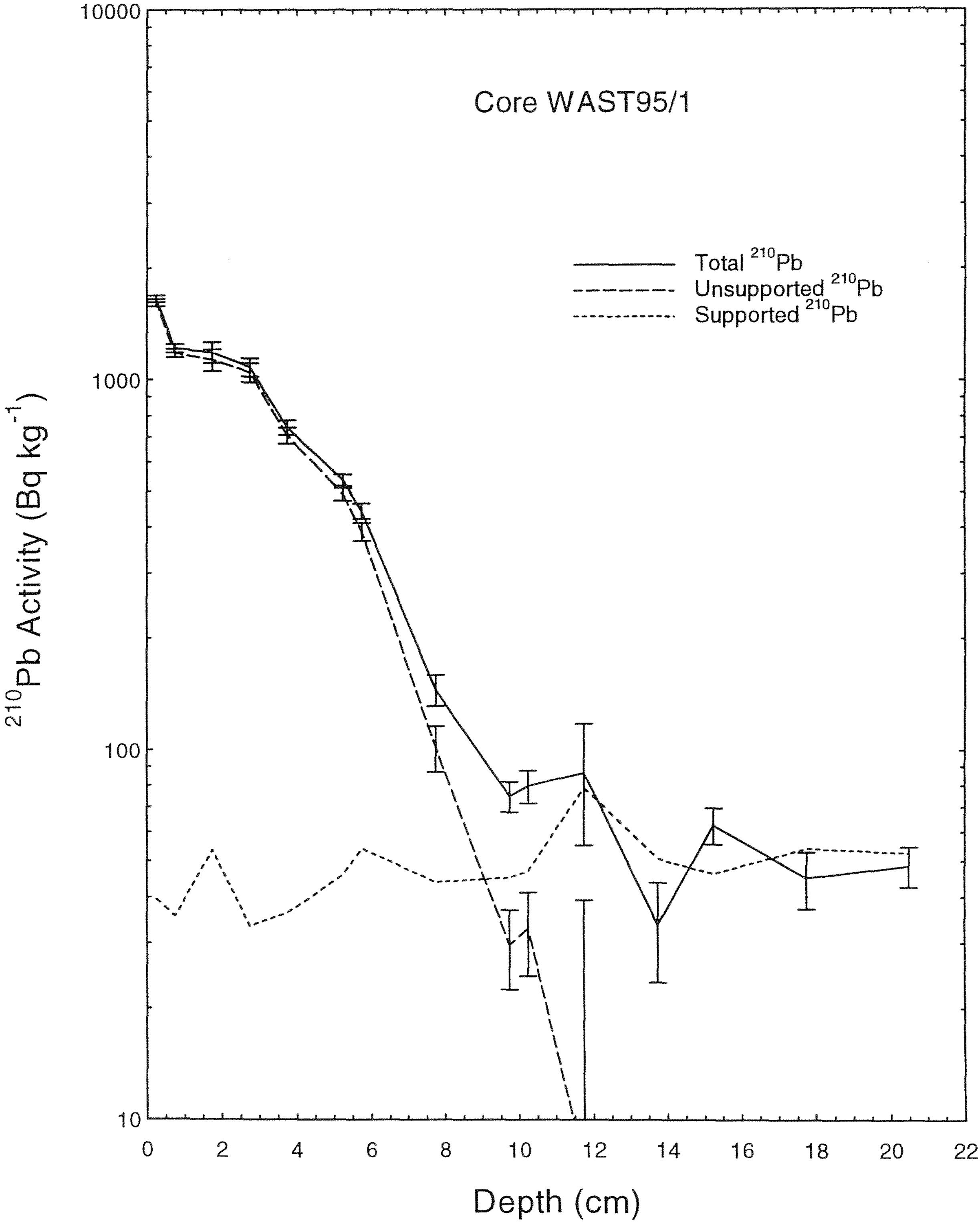
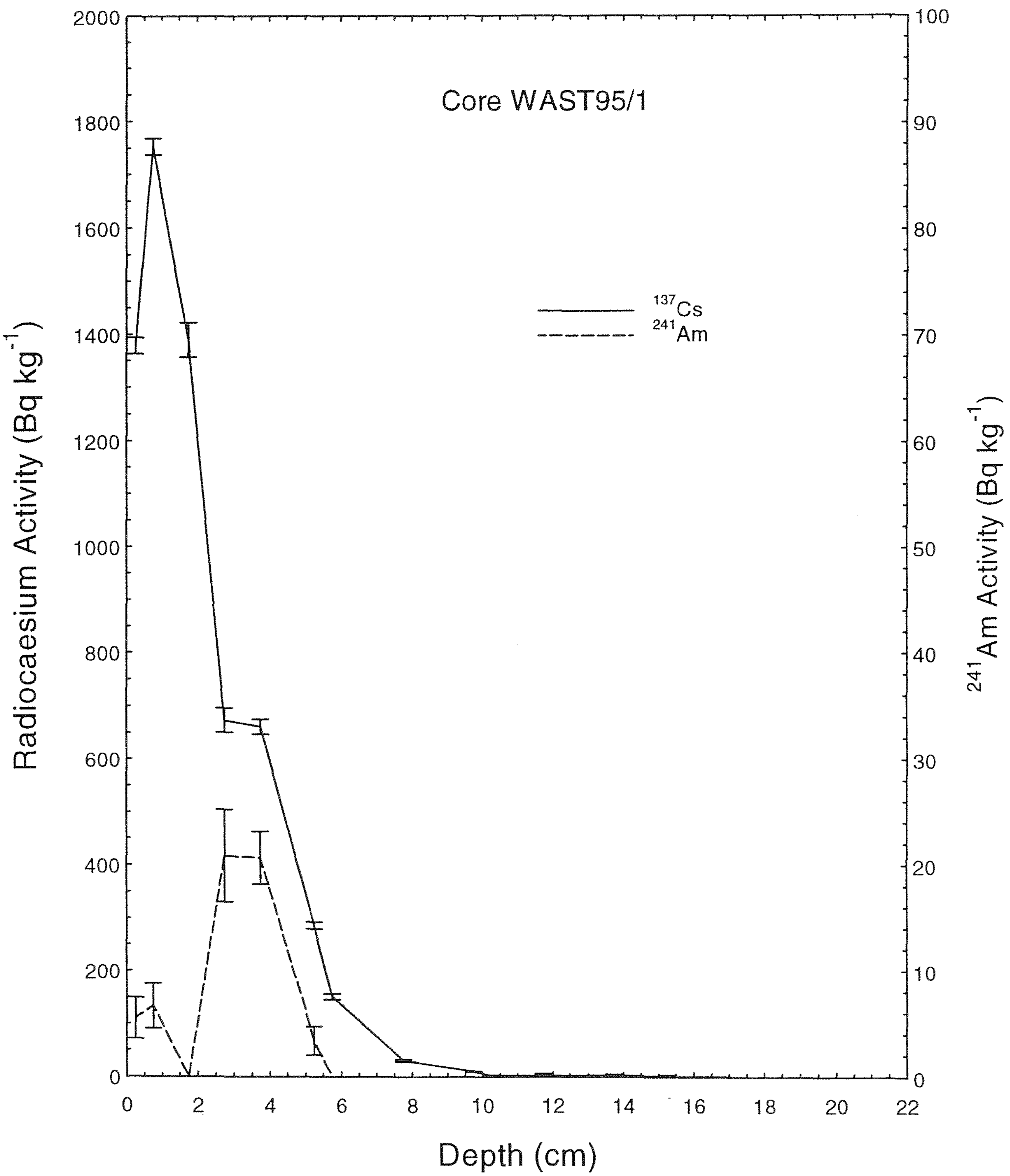


Figure 9 ^{137}Cs and ^{241}Am Activity versus Depth - Wastwater



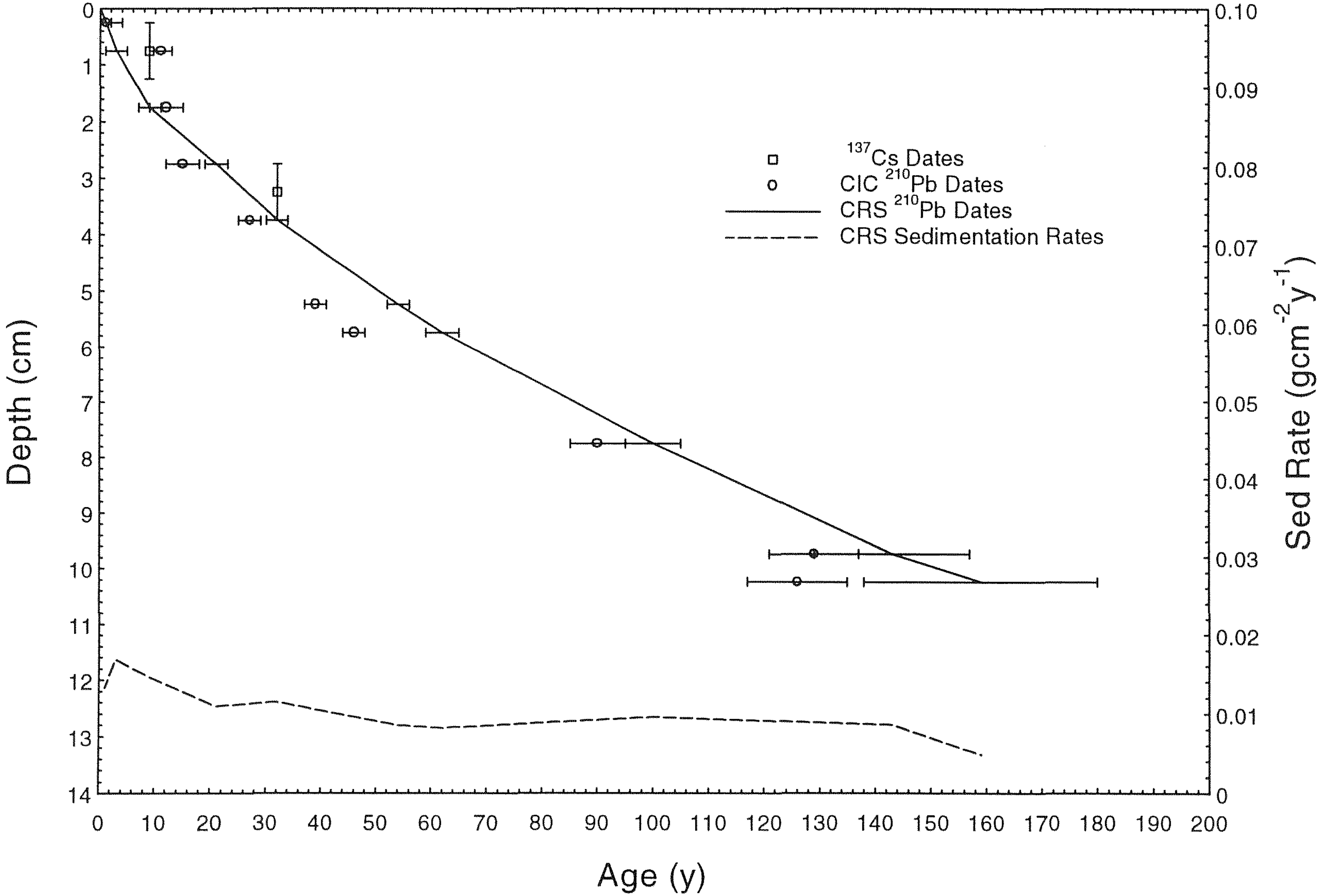
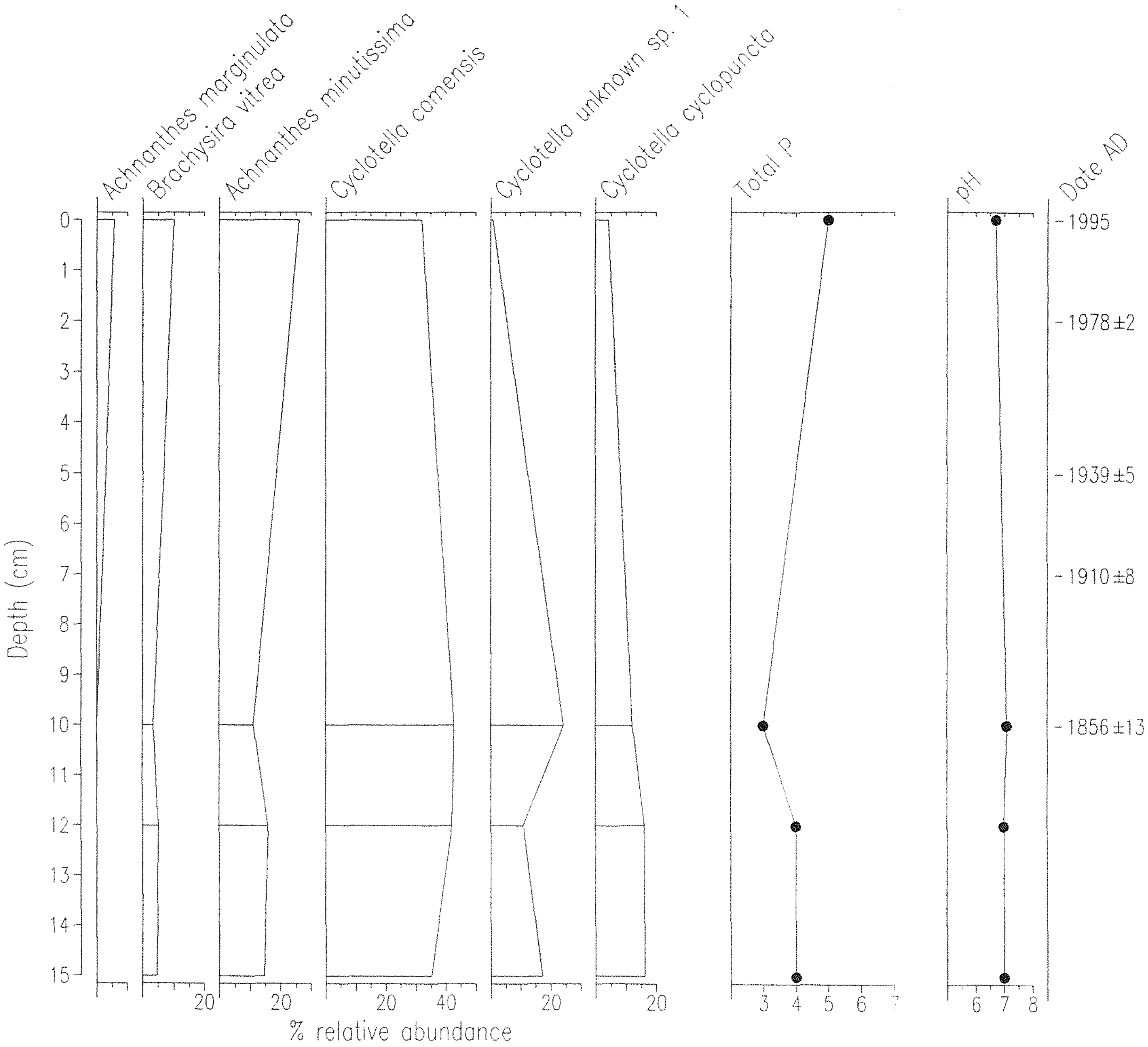


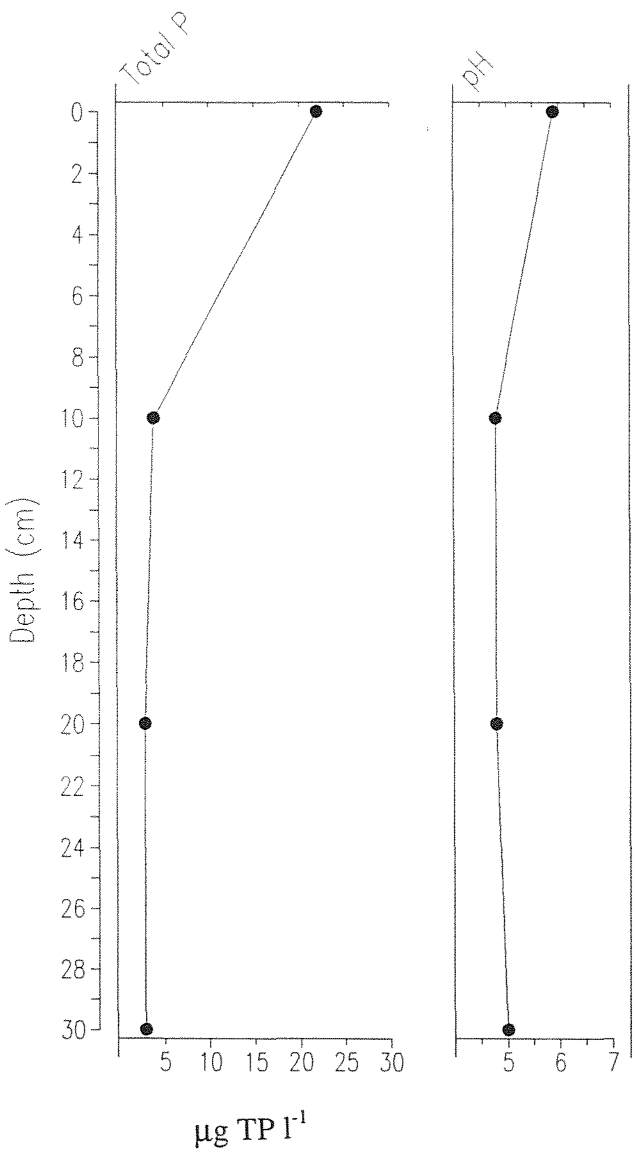
Figure 10 Depth versus Age - Wastwater

Figure 11 Summary diatom diagram and reconstructions for Wastwater



Oak Mere

- Oak Mere is a unique site with a complex hydrology and therefore the diatom models cannot be applied with any confidence. The results should be interpreted with caution.
- However, the diatom flora does suggest that the lake has always been acid but has become more base-rich/nutrient-rich in recent years.
- The species changes are more likely to be responses to changes in pH rather than to changes in nutrient concentrations, and may represent natural fluctuations between acid and alkaline conditions.



6. Oak Mere, Cheshire (SJ 574 677)

6.1 Site Description and Water Chemistry

- Altitude 72 m
- Area 22 ha
- Catchment area 2.56 km^2
- Mean depth 2 m
- Maximum depth 8 m
- The lake is occasionally fed by one small inflow channel at the NW of the site but is largely fed by rainwater, and to some extent by groundwater and springs.
- The site lies close to fault separating Keuper sandstone from the Keuper marl, and is overlain by glacial sands and gravels, and by peat in the N.
- The land use is principally improved agriculture, with some woodland and heath at the NW end. Sand extraction takes place to the N of the lake.

pH	units	5.1
conductivity	$\mu S\ cm^{-1}$	187
alkalinity	$meq\ l^{-1}$	0.03
chlorophyll a	$\mu g\ l^{-1}$	7.4
nitrate	$mg\ l^{-1}$	0.2
inorganic nitrogen	$mg\ l^{-1}$	0.3
total phosphorus	$\mu g\ l^{-1}$	61

6.2 Macrophyte Survey

Date of visit 4-7-96.
The poor aquatic macrophyte species diversity of Oak Mere has been noted on a number of recent surveys and probably reflects the low pH (mean pH = 5.1) and the sandy nature of the littoral which is unsuitable for the establishment of many species. A map of the aquatic macrophyte distribution of Oak Mere is presented in Figure 12. Water turbidity was poor on the day of the survey and a secchi disc depth of 2.3 m was recorded.

The submerged flora largely consists of *Littorella uniflora* which occurs in shallow water around much of the perimeter, occasionally in association with *Juncus bulbosus* var. *fluitans*. *Scirpus fluitans* is also present at the south end of the lake. The encircling shoreline community is dominated by *Juncus effusus* and *Eleocharis palustris* where *Hydrocotyle vulgaris* is occasional. Two small stands of *Typha latifolia* are present in sheltered locations at the north end of the site. Small patches of *Callitriche* sp. also occur at the north end.

There is some evidence to suggest recent small recent changes in the composition of the macroflora at Oak Mere. In recent years aquatic macrophytes surveys of Oak Mere have been undertaken by the Nature Conservancy Council in 1978 (Newbold and Walker) and 1979. In both surveys the aquatic flora was found to include several bryophytes. *Sphagnum auriculatum* and *Drepanocladus fluitans* were found in fully submerged locations while *Atrichum crispum* and *Pohlia bulbifera* were recorded as frequent in littoral areas on the east side. However no fully submerged aquatic bryophytes were found during this survey. There was no evidence either, in this survey, of the small patch of *Nuphar lutea* recorded on most recent surveys. It is likely that the littoral cover of *Juncus bulbosus* var. *fluitans* has increased since the earlier surveys. Recent work in the Netherlands has shown that growth of *J.bulbosus* in oligotrophic pools is encouraged by enhanced nitrogen levels (e.g. Schuurkes *et al.* 1987). However, *S. auriculatum* and *D. fluitans*, species which appear to have declined at Oak Mere, are also shown by the Dutch studies to benefit from nitrogen enrichment and therefore the causes of possible *J.bulbosus* expansion remain unclear.

Using the species list in Table 4, Oak Mere is classified as Type 2 (Palmer, 1992), an oligotrophic category.

Table 4 Species list and DAFOR abundance rating for Oak Mere

	abundance	notes
Submerged taxa		
Littorella uniflora	A	along most of littoral to 40 cm water depth
Juncus bulbosus var. *fluitans*	O	usually in association with Littorella uniflora
Callitriche sp.	R	small patches at n end
Emergent taxa		
Eleocharis palustris	O	locally abundant in shallow water
Hydrocotyle vulgaris	F	
Juncus effusus	A	surrounding most of the lake perimeter
Typha latifolia		two small stands in n end

6.3 Lithostratigraphy

An 84 cm sediment core (OAKM1) was taken from the deepest part of the lake in a water depth of 7.7 m using a Mackereth corer on 26-1-95. The lower sediments (> 40 cm) were a very dark brown (10YR 2/2), silty lake mud with plant remains and traces of sand (Ld2 Ag1 Dh1 Dg+ Ga+). There was a distinct change at 40 cm to a dark greyish brown (10YR 4/2) silt and coarse sand layer with gravel (Ag2 Ga1 Gs1 Gg+ Ld+) between 30-40 cm. Above this layer, the sediment was very dark brown again with similar composition to the lower core section (Ld2 Ag1 Dh1 Lso++ Dg+). Diatoms were only present in the upper 30 cms. There was another sandy layer at c.8 cm, above which lay a more organic layer (3-8 cm). The upper few cms were a very dark greyish brown (10YR 3/2), fine sand and silty mud (Ld1 Lso1 Ga1 Ag1 Dh++).

The %dw and %loi profiles (Figure 13) show marked changes in the organic content of the sediment. In the lower 40 cms, %dw was c.15-20% and %loi c.30-45%. There was a distinct layer of minerogenic sediments, possibly marking an inwash episode, between 30-40 cm, where %dw increased sharply to 70% and %loi decreased to only 2%. Above 30 cm, the values reverted to those similar to the lower core section. At c.8 cm, there was another marked decline in organic content with values similar to those in the 30-40 cm section, followed by a layer of higher %loi values (40%) from 3-8cm. The surface was highly inorganic with %dw of 60% and %loi of 4%.

6.4 Radiometric Dating

This core does not appear to contain a coherent record of fallout radionuclides and it was not possible to make any reasonable estimate of sedimentation rates. In all samples analysed (from the top 20 cm) ^{210}Pb activities were essentially in equilibrium with the supporting ^{226}Ra (Figure 14). Significant traces of ^{137}Cs were detected only in the topmost sample (Figure 15). The difficulties encountered with the core may be related to the events giving rise to the layers of unusually dense sediment in the top 3 cm, at 7-8 cm, and at 32-39 cm.

6.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in four levels of the sediment core were calculated and Figure 16 illustrates the results for the major taxa. Diatom preservation was very poor throughout the core, except in the surface sample, and there were no diatoms below 30 cm. A total of 64 taxa was observed, 41 of which were present in the TP calibration set and 35 of which were present in the pH calibration set. There were particularly bad analogues for the TP calibration, owing to the dominance of acidophilous taxa that were not present in the northwest European TP data set. Less than 70% of the fossil assemblage could be used in the TP calibration procedure. Analogues were better for the pH reconstructions with over 85% of the fossil assemblage being used in the pH calibration procedure, except in the surface sample where only 64% could be used.

Figure 16 illustrates that there have been marked changes in the diatom species composition over the period represented by the core. The unusually dense sediment in the upper 3 cms, at c.8 cm and at 30-40 cm caused difficulties for radiometric dating and there was no coherent record of fallout radionuclides, thus a chronology for the core could not be determined. The origin of these mineral layers is unknown, although fluctuations in water level, related to changes in net precipitation and addition of borehole water have been well documented (Savage *et al.*, 1992) and sand extraction is known to occur in the catchment.

The two lower samples (30 cm and 20 cm) were almost identical, dominated by acidophilous, non-planktonic taxa, particularly *Frustulia rhomboides*, *Eunotia [cf. minima]*, *Cymbella perpusilla*, *Eunotia incisa*, *Eunotia rhomboidea* and *Fragilaria virescens* var. *exigua*. There was a slight change by the 10 cm sample where *Eunotia rhomboidea* and *Fragilaria virescens* var. *exigua* disappeared, and *Navicula [cf. seminulum]* and *Fragilaria construens* var. *venter*, appeared for the first time, the latter two taxa being associated with more alkaline waters. *Pinnularia viridis* also increased in importance. The surface sample was markedly different from the 10 cm sample. The acidophilous taxa decreased in relative abundance, particularly *Eunotia [cf. minima]* and *Frustulia rhomboides*. Conversely, the relative abundances of the more circumneutral to alkaliphilous taxa *Fragilaria construens* var. *venter* and *Navicula [cf. seminulum]* increased, and *Surirella linearis* was observed for the first time.

6.6 Total Phosphorus and pH Reconstructions

The TP reconstruction shows that there have been changes in the nutrient status of Oak Mere. Given the analogue and dissolution problems, however, the diatom-inferred results are not reliable and only a qualitative interpretation of the species changes can be made. The values produced by the model ranged from 3 $\mu g\ l^{-1}$ for the 30 cm and 20 cm samples, to 4 μg TP l^{-1} for the 10 cm sample, increasing to 22 μg TP l^{-1} for the 1995 sample. The higher value for the surface sample is explained by the higher relative abundance of *Fragilaria construens* var. *venter*, which has an intermediate TP optimum, and the reduced importance of the acid-tolerant taxa that dominated the lower core levels.

The pH reconstruction results must also be viewed in light of the analogue difficulties, especially for the surface sample. The diatom-inferred pH values for the three lower levels were very similar at c. 4.9, increasing to 5.9 for the 1995 sample.

6.7 Discussion

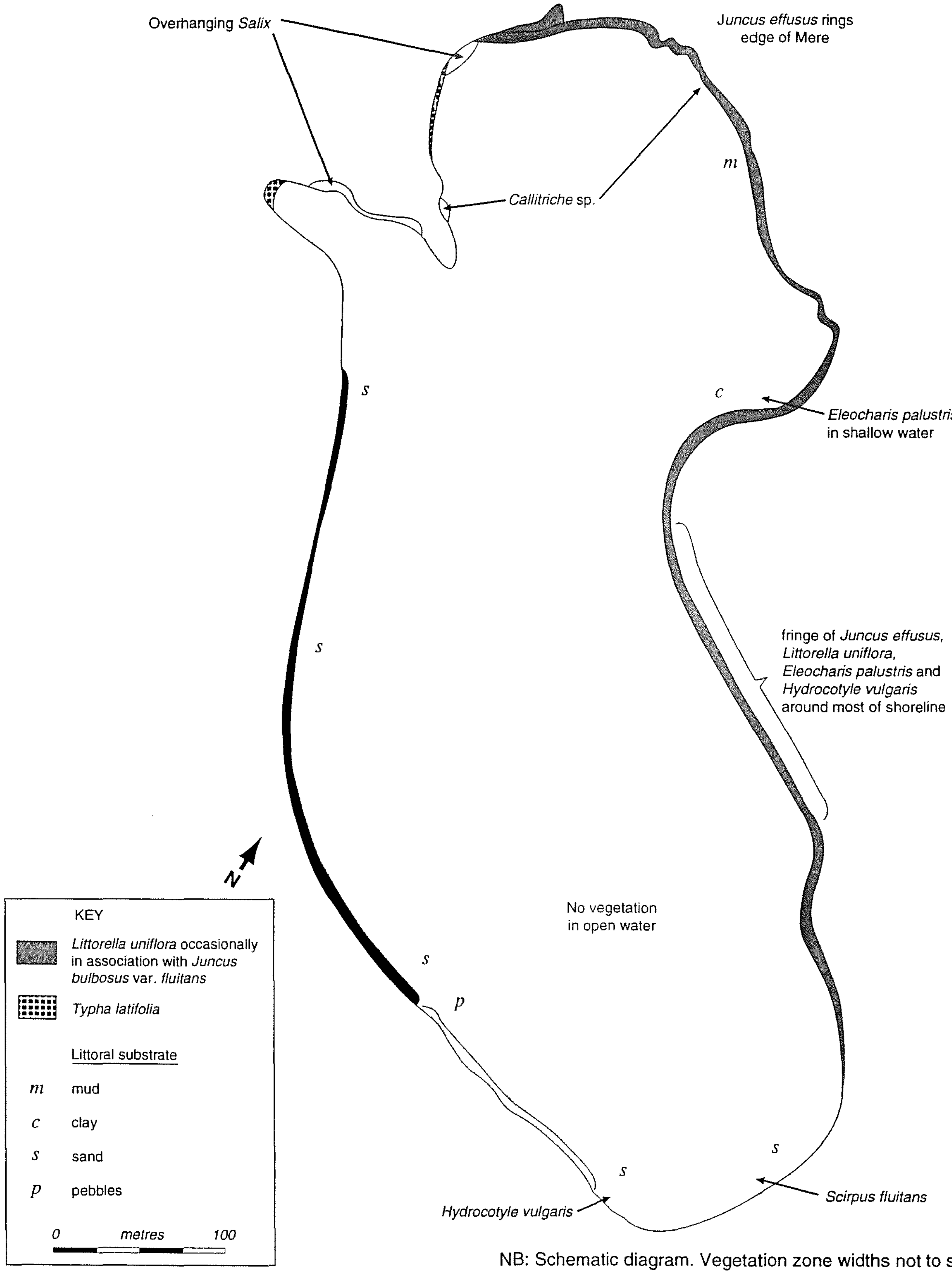
Oak Mere is a unique lake because of its low pH, alkalinity and conductivity and yet relatively high nutrient concentrations. It is a lowland acid lake, which contrasts sharply with the other meres of the Cheshire-Shropshire Plain. Unfortunately there are no similar lakes in the northwest European TP and pH calibration sets and therefore the diatom models cannot be used with any confidence to reconstruct the lake's history. There are also the additional problems associated with dating the core and a chronology could not be established. The diatom flora, however, does suggest that the lake has always been acid with dominance of acidophilous (low-pH tolerant) species throughout the core. The reconstructed pH values compare favourably with measured data for 1991/2 where mean pH was 5.1 and pH ranged from 4.5-5.7 over the sampling period (Moss *et al.*, 1992). The diatoms in Oak Mere (as in Clarepool Moss) appear to be responding more to the low pH of the water column than to nutrient concentrations.

The pH model indicates that the lake has become more base-rich in recent years. Hydrological studies of Oak Mere have reported that there are at least two separate sources of water to the lake; a base-poor surface supply entering via superficial permeable drift and a base-rich supply entering through partially or intermittently permeable lake sediments (Savage *et al.*, 1992). This study provided evidence for natural variation of the waters between acid base-poor and moderately base-rich conditions, although the latter state has also occurred following sporadic additions of borehole water. Oak Mere has a history of water level fluctuation. Water was also abstracted from the mere between 1926 and 1948, but a study over the period 1990-1992 (Carvalho, 1993) showed that water levels may still be falling. Given the very complex hydrology of the mere, it is difficult to interpret changes in water quality and the observed diatom species changes in the core may simply reflect natural fluctuations between acid and alkaline conditions.

The reconstructed TP values are clearly under-estimates of the true TP concentrations of the lake because of the lack of species analogues. Measured TP data gives an annual mean of 61 μg TP l^{-1} and a seasonal range of 30-99 μg TP l^{-1} (Moss *et al.*, 1992) compared to a current DI-TP value of 22 μg TP l^{-1}. The TP concentrations of Oak Mere are higher than might be expected for a lake of comparable conductivity and alkalinity in an upland catchment but are low relative to many other of the North-West Midland Meres (Moss *et al.*, 1992). The model indicates that there has been nutrient enrichment at Oak Mere and this might be expected given the agricultural intensification in the Cheshire region. Moss *et al.* (1992) reported that total nutrient units in the lake catchment had increased between 1931 and 1987. There is also local anecdotal evidence of increasingly frequent algal blooms since the late 1980s but it is not clear whether this is an indication of enrichment or due to changes in lake calcium and pH (Reynolds & Allen, 1968). The nutrient signal is also complicated by the fluctuations in water level in the mere. A two year study by Carvalho (1993) established a significant correlation between decreasing nutrient concentrations in the mere and the declining water level associated with drier years.

In summary, Oak Mere is a unique site with a complex hydrology and it is therefore difficult to place it within the trophic classification scheme for freshwaters. It is often described as an oligotrophic lake because of its low pH and alkalinity and its low algal crops but in terms of its TP concentrations, it would be classed as a eutrophic lake. This atypicalness implies it would be difficult to use Oak Mere as an example lake for providing restoration targets for other UK sites.

Figure 12 Aquatic macrophyte distribution map for Oak Mere



NB: Schematic diagram. Vegetation zone widths not to scale.

Figure 13 Lithostratigraphic data for Oak Mere

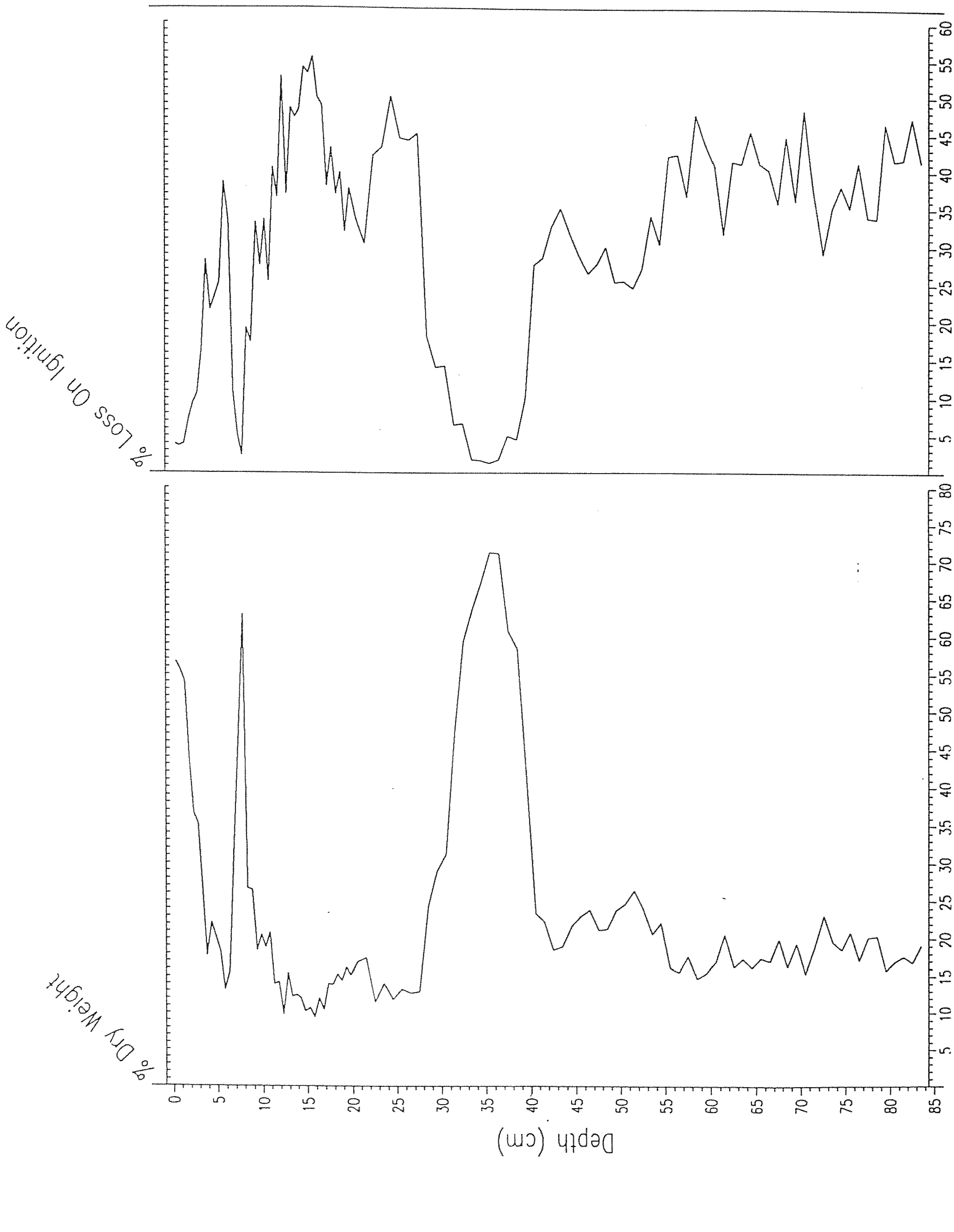


Figure 14 ^{210}Pb Activity versus Depth - Oak Mere

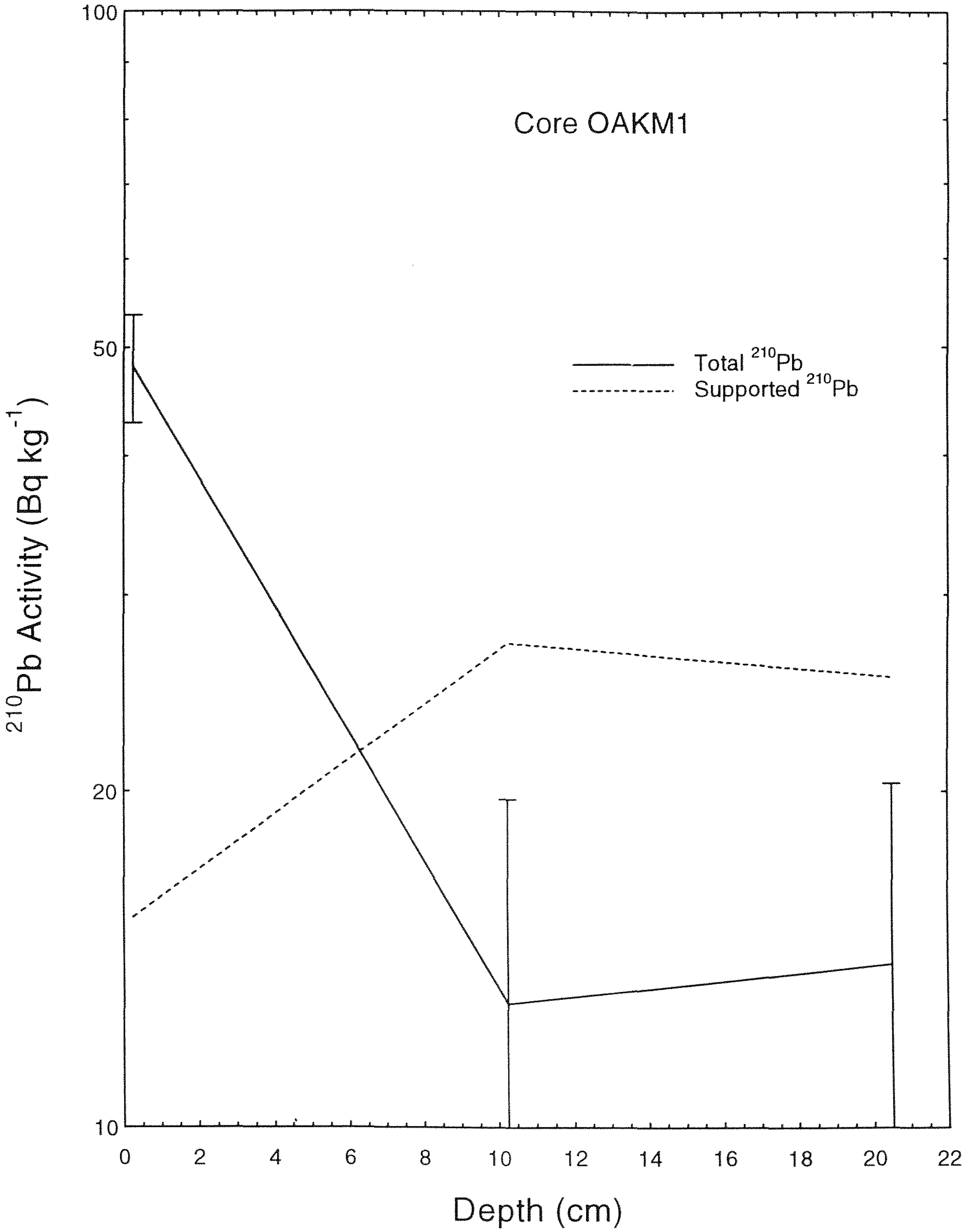


Figure 15 ^{137}Cs Activity versus Depth - Oak Mere

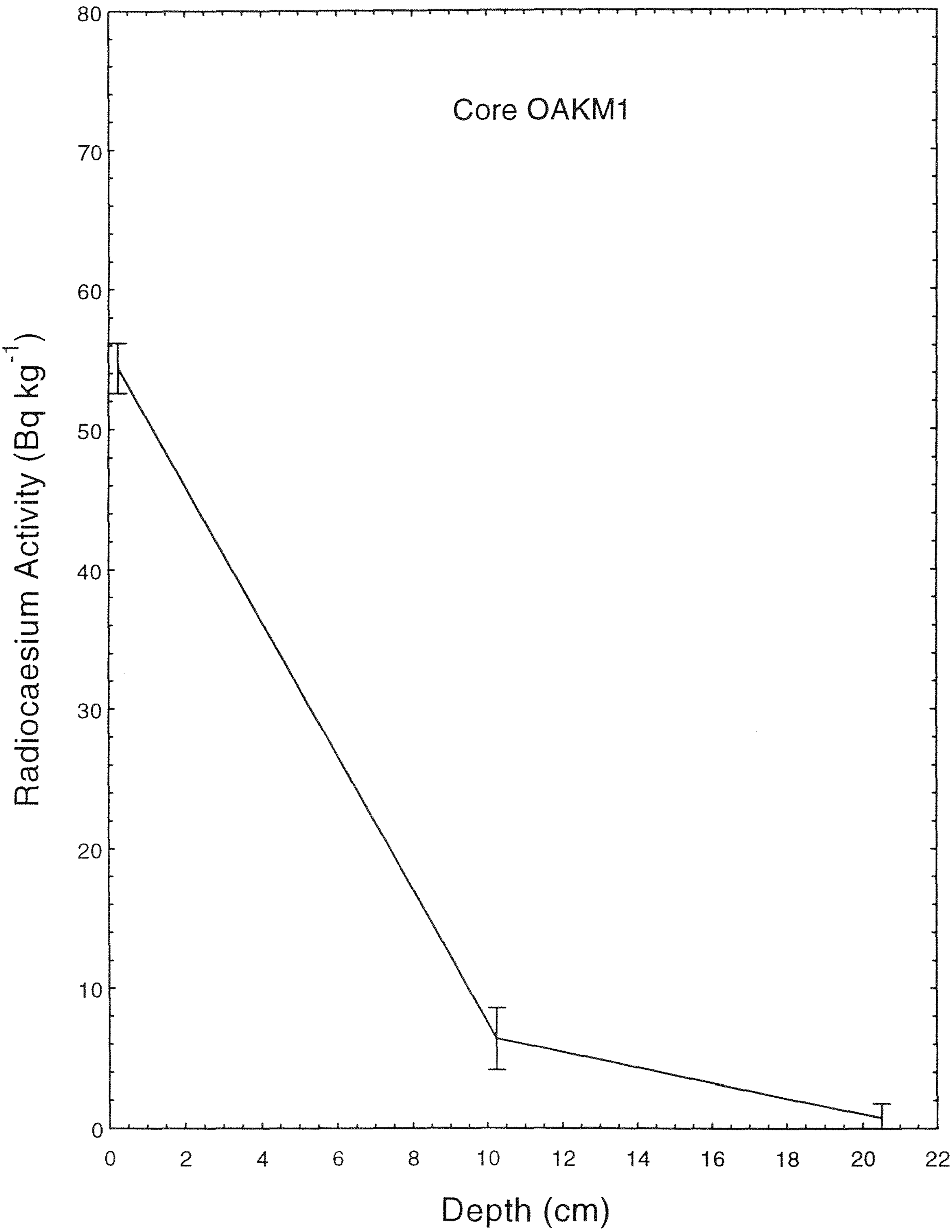
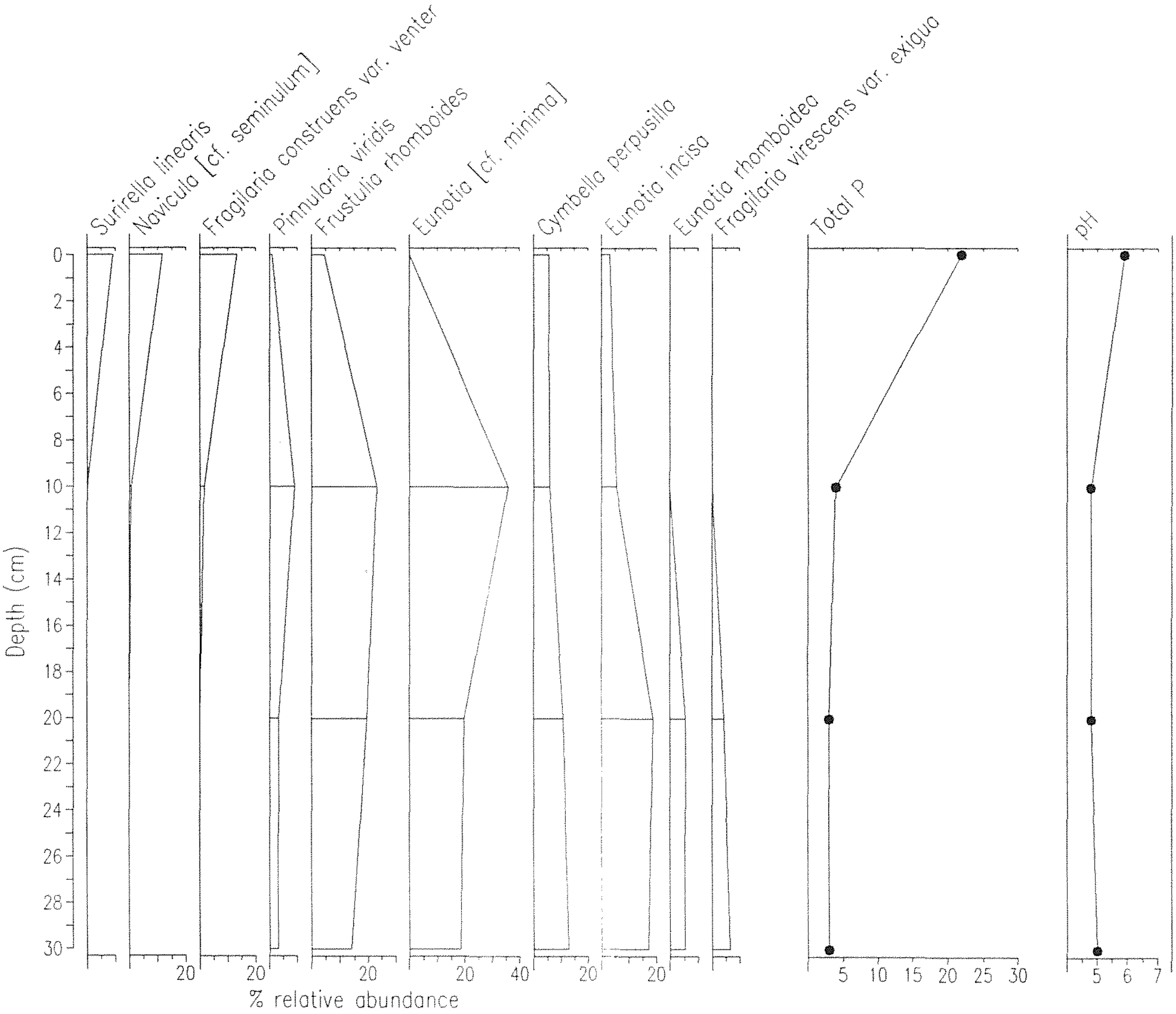
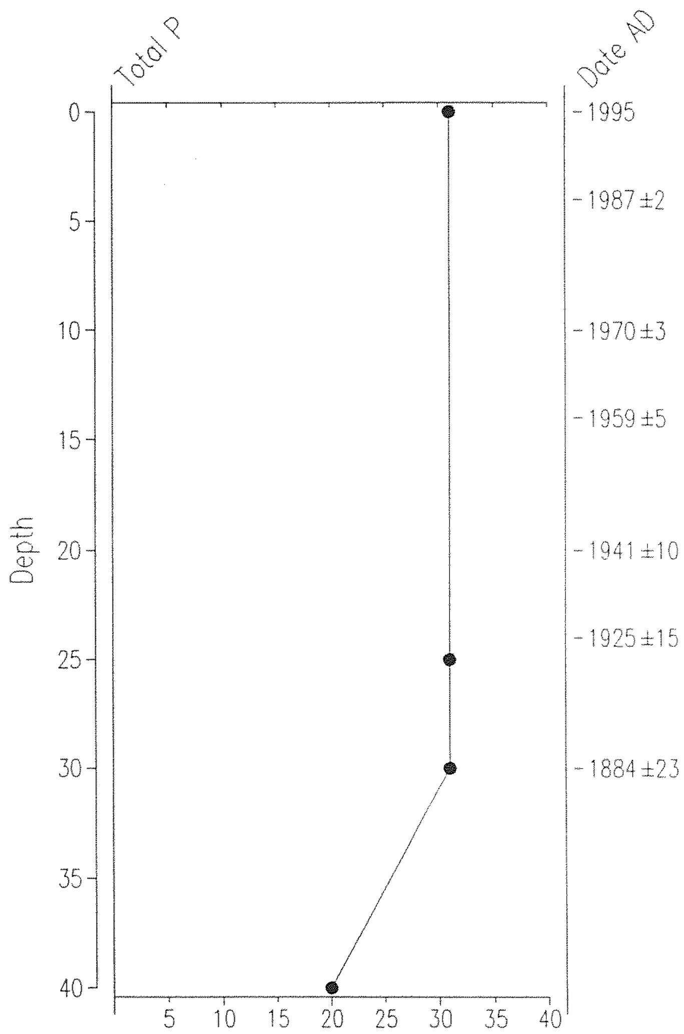


Figure 16 Summary diatom diagram and reconstructions for Oak Mere



Bassenthwaite Lake

- Bassenthwaite Lake represents a typical mesotrophic lake in a near natural condition.
- However, diatom species shifts were observed, indicating ecological change in recent years and suggesting a need for continued monitoring of the lake.



µg TP l^{-1}

7. Bassenthwaite Lake, Cumbria (NY 214 296)

7.1 Site Description and Water Chemistry

- Altitude 75 m
- Area 5.28 km^2
- Mean depth 5.3 m
- Maximum depth 21 m
- Flushing rate 30 days
- Approximately 29% of the total drainage area is cultivable; forestry is important in the catchment.
- Large mountainous catchment area of 238 km^2

pH	units	7.3
conductivity	$\mu S\ cm^{-1}$	74
alkalinity	mg $CaCO_3$	11
calcium	mg l^{-1}	6.3
total nitrogen	mg l^{-1}	0.8
total phosphorus	$\mu g\ l^{-1}$	25

nb. These readings are taken from the lake outflow (at Ouse Bridge)

7.2 Macrophyte Survey

Date of visit 6/7-7-96.
The aquatic macroflora of Bassenthwaite Lake reflects a mesotrophic water body, incorporating species characteristic of lakes of circumneutral pH and relatively low nutrient status. Despite the relatively shallow nature of the lake, the aquatic macrophyte communities are generally restricted to locations with a maximum depth of approximately 2.5 m and therefore close to the shoreline, as a result of the water turbidity. Maps of the aquatic macrophyte distribution of Bassenthwaite and transect profiles are presented in Figure 17, Figure 18, Figure 19, and Figure 20.

Floristically, the lake habitats can be divided between the steeply shelving and exposed west side of the lake and the east side and north and south ends which are more gently sloping and include a number of sheltered bays. During the current survey, the west side was not thoroughly examined. Its steeply shelving boulder dominated shoreline is not conducive to macrophyte colonisation although it is possible that deep water vascular species e.g. *Isoetes lacustris* may occur in patches offshore.

Sheltered parts of the shoreline in the north-west end of the lake provide habitats for a number of species. Several small stands of *Scirpus lacustris* spp. *lacustris* occur here, usually in association with *Littorella uniflora.* A diverse association of the latter with *Equisetum fluviatile*, *Potamogeton perfoliatus*, *Ranunculus flammula*, *Caltha palustris*, *Eleocharis palustris* and occasional *Apium nodiflorum* occurs near Dubwath, while *Isoetes lacustris* and *Nitella flexilis* var. *flexilis* are present in deeper water. In another nearby shallow water location protected by a stand of *S. lacustris* ssp. *lacustris* and dominated by *E. palustris*, *Alisma plantago-aquatica*, *Polyganum amphibium*, *Nuphar lutea*, *Lythrum portula* and *Callitriche hermaphroditica* were locally frequent.

The north end of Bassenthwaite Lake supports a shoreline association of *C. palustris*, *R. flammula*, *Mentha aquatica* and *Myosotis* sp. on the open water side of an almost continuous sward of *Phalaris arundinacea. L. uniflora* is the only submerged species in shallow water here and much of the littoral zone is dominated by a wave washed cobble substrate. However *I. lacustris* is present at around 2.5 m water depth. The length of shoreline between Armathwaite and Scorness bay includes several shallow, sheltered habitats including a small bay which supports an association of *Elatine hexandra*, *Ranunculus peltatus* and *Myosotis* sp.. *Myriophyllum alterniflorum* and *L. uniflora* are present in deeper water, and *P. perfoliatus* becomes more frequent towards Scorness bay.

The shallow water of Scorness bay is dominated by *L. uniflora*, *M. alterniflorum with P. gramineus* locally frequent. *I. lacustris* and *N.flexilis* are abundant between 1.0 and 2.5 m water depth.

Bowness Bay contains a particularly diverse floral assemblage, with several stands of floating leaved, emergent and submerged macrophyte species present (see Figure 19). A large stand of *Potamogeton natans* in the centre of the bay is flanked by stands of *Equisetum fluviatile*, *Nuphar lutea* and *Polyganum amphibium*, while two discreet stands of *S. lacustris* ssp. *lacustris* are present adjacent to the shoreline. A few individuals of *Nymphaea alba* occur amongst the *N. lutea* stand. The heavily grazed southern shoreline of the bay is mainly dominated by *L. uniflora* with *M. alterniflorum* and *P. gramineus* is locally frequent. *Elodea canadensis* forms a dense sward in deeper water although this is interrupted by a stand of *P. perfoliatus.*

Church Bay contains few plants in an open water habitat although *Callitriche hamulata* and *Fontinalis antipyretica* are present close to the shore. The shoreline between Church Bay and Little Crowsthwaite is exposed, bolder dominated and largely devoid of macrophytes although a dense stand of *N. flexilis* var. *flexilis* and *F. antipyretica* occurs out from the inflow at Little Crowsthwaite.

The large bay at the south end of the lake is shallow and sandy and dominated by *F. antipyretica*, *N. flexilis* var. *flexilis* and *Elodea nutallii.* One fragment of *Potamogeton crispus* was recovered from a double-headed rake trawl from this bay, the only observation of this species during the current survey. The shoreline is fringed by stands of *S. lacustris* var. *lacustris*, *Phragmites australis*, *Carex* sp.and on the south side *Alnus* sp., with occasional stands of *N. lutea*. *E. canadensis*, *A. plantago-aquatica* and *R. flammula* are also locally frequent in the more sandy areas. The lagoons lying in shingle south of Blackstock Point were not included on this survey.

The species list compiled for this survey (Table 5) is broadly similar to those provided by Stokoe (1980) and Newbold and Palmer (1981), the main exceptions being the absence of records for *Potamogeton alpinus* and *Potamogeton pusillus* which were recorded in 1980 and 1981. In addition *Apium inundatum* which was recorded as rare in the 1981 survey was not found.

P.pusillus was recorded as rare in the 1980 survey and locally abundant in the north end of the lake in the 1981 survey. *P. alpinus* was found in local abundance at one location only in both surveys. The latter species is widespread in lakes and rivers in northern Britain where there is little evidence of decline in recent years as a consequence of anthropogenic impacts (Preston and Croft, 1997).

Applying the species list presented below, Bassenthwaite Lake is classified as Type 5a (Palmer, 1992), a mesotrophic category.

Table 5 Species list and DAFOR abundance rating for Bassenthwaite Lake

	abundance	notes
Submerged taxa		
Callitriche hamulata	R	in s end and s of Bowness Bay
Callitriche hermaphroditica	O	in n-w
Elatine hexandra	O	in n-e
Elodea canadensis	O	in
Elodea nutallii	O	Bowness Bay and s end
Fontinalis antipyretica		s of Bowness Bay and s end
Isoetes lacustris	A	dominant between 1.5 m and 2.5 m depth n end to Bowness Bay
Littorella uniflora	A	numerous locations, most abundant n end
Lythrum portula	R	n end
Myriophyllum alterniflorum	F	e side in large bays
Nitella flexilis var. *flexilis*	F	several locations to 1.5 m depth
Potamogeton berchtoldii	R	detached fragment only found, e side
Potamogeton crispus	R	single fragment recovered from rake trawl s end
Potamogeton gramineus	O	in large bays on e side in shallow water
Potamogeton perfoliatus	F	locally dominant to 1.5 m water depth
Ranunculus peltatus	R	n end
Floating leaved taxa		
Lemna minor	R	in n-w
Nuphar lutea	O	sheltered locations throughout
Nymphaea alba	R	in Bowness Bay
Polyganum amphibium	R	in n-w
Potamogeton natans	R	in Bowness Bay
Emergent taxa		
Apium nodiflorum	R	
Caltha palustris	F	
Carex rostrata	R	
Carex vesicaria	O	
Eleocharis palustris	A	
Equisetum fluviatile	O	
Juncus effusus	F	
Mentha aquatica	F	
Phragmites australis	R	
Phalaris arundinacea	A	
Ranunculus flammula	F	
Scirpus lacustris ssp. *lacustris*	F	

7.3 Lithostratigraphy

An 80 cm sediment core (BASS1) was taken from the deepest part of the lake at a water depth of 20 m using a Mackereth corer on 12-6-95. The lower part of the sediment core (30-70 cm) was a dark olive grey (5Y 3/2), silty lake mud with clay and plant fragments (Ag2 Ld2 As+ Lso+ Dh+). One notable feature was a very dark greyish brown (2.5Y 3/2) layer at 26-27 cm with a high clay-silt content (Ld2 Lso1 Ag1 As+ Dh+). The origin of this layer is uncertain (see chronology section).There was a colour change above this layer to a dark olive brown (2.5Y 3/3) although sediment composition remained largely unchanged. The upper 5 cms were a dark brown (10YR 3/3), organic lake mud (Ld3 Lso1 Dg+ Ag+).

The %dw and %loi profiles (Figure 21) show that %dw fluctuated c.40% and %loi c.9% in the lower section of the core (35-80 cm). There was a progressive increase in percentage organic matter from 35 cm to the core top, %dw falling to 10% and %loi rising to 15% at the sediment surface. The clay layer was apparent at 26 cm with a marked increase in %dw to 43% and a corresponding decrease in %loi to 5%. The changes in wd were similar to those in %dw with values of c. 1.3 g cm^{-3} from 35-80 cm and then a decrease from 35 cm to the core top where values were only 1.1 g cm^{-3}. The clay layer was not detected in the wd profile because of the low resolution of the wd measurements (every 5 cm).

7.4 Radiometric Dating

The Bassenthwaite Lake core had a good ^{210}Pb record, closely matching that of a recent (1994) core from the same lake. In both cores equilibrium with the supporting ^{226}Ra was reached at a depth of c.32 cm. In the 1994 core the CRS model ^{210}Pb dates were validated by well resolved ^{137}Cs and ^{241}Am peaks defining the 1986 (Chernobyl) and 1963 (weapons test fallout) levels. Although the restricted number of measurements did not allow precise resolution of these features in the 1995 core, the results obtained were sufficient to suggest the two cores also had closely matched artificial radionuclide records. There was again good agreement with the CRS model dates, and this has been used to construct the chronology given in the following table. As in the 1994 core, the results indicated a significant increase in sedimentation rates during the period 1900-40 that has been more or less sustained to the present day.

The ^{210}Pb record terminates at 30 cm (dated c.1880), just above a characteristic clay layer marked by much higher sediment densities. The origin of this layer is not certain, but possible causes include mining, railway construction (c.1855) or construction of the Thirlmere dam (c.1890). In view of this uncertainty it would be inappropriate to extrapolate the chronology below 30 cm.

The results are shown in Table 6 and Figure 22, Figure 23 and Figure 24.

Table 6 Chronology of Bassenthwaite Lake

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	$g\ cm^{-2}$	AD	y	±	$g\ cm^{-2}\ y^{-1}$	$cm\ y^{-1}$	± (%)
0.0	0.00	1995	0				
2.0	0.31	1992	3	2	0.098	0.51	13.1
4.0	0.75	1987	8	2	0.095	0.40	13.9
6.0	1.27	1982	13	2	0.097	0.33	15.8
8.0	1.90	1976	19	3	0.112	0.36	19.4
10.0	2.54	1970	25	3	0.126	0.38	23.0
12.0	3.28	1964	31	4	0.141	0.39	26.3
14.0	4.01	1959	36	5	0.122	0.34	28.4
16.0	4.73	1953	42	6	0.116	0.32	31.6
18.0	5.45	1947	48	8	0.137	0.37	36.6
20.0	6.21	1941	54	10	0.113	0.30	43.4
22.0	6.96	1933	62	12	0.097	0.26	54.3
24.0	7.70	1925	70	15	0.083	0.22	65.7
26.0	8.48	1914	81	18	0.069	0.18	76.5
28.0	9.31	1901	94	20	0.053	0.13	86.3
30.0	10.11	1884	111	23	0.041	0.09	97.0

7.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in four levels of the sediment core were calculated and Figure 25 illustrates the results for the major taxa. Diatom preservation was good to 30 cm downcore but was poor in the more inorganic, lower core section. A total of 97 taxa was observed, 77 of which were present in the calibration set. All of the common taxa were well represented in the calibration set, with greater than 90% of the fossil assemblage being used in the calibration procedure.

Figure 25 illustrates that there has been a slight change in the diatom species composition over the period represented by the upper 40 cm of the core (c.1850-1995). The bottom sample (40cm: pre-1880) was dominated by *Achnanthes minutissima* and *Synedra nana*, species generally found in oligo-mesotrophic waters. The 30 cm sample (c.1880) also had a high relative abundance of *Achnanthes minutissima* (30%) and *Synedra nana* (40%), and *Asterionella formosa*, a planktonic species typical of mesotrophic waters, increased in importance. The 25 cm level (c.1920) was similar to the 30 cm sample except for the higher frequency of *Fragilaria crotonensis*, a planktonic diatom also associated with mesotrophic waters. Two taxa, *Aulacoseira subarctica* and *Cyclotella pseudostelligera* appeared only in the surface sample (1995) but the relative abundances of the other major taxa remained more or less unchanged.

7.6 Total Phosphorus Reconstruction

The TP reconstruction shows that the lake has been mesotrophic for the whole of the post 1850 period with only a slight increase in TP concentrations. The model provided an estimate of 20 µg TP l^{-1} for pre-1880 (40 cm), and then stabilised with a diatom-inferred value of 31 µg TP l^{-1} for 1880 (30 cm), 1920 (25 cm) and 1995. This indicates that there was a slight enrichment in the late nineteenth century but that there was no evidence of any further recent eutrophication.

7.7 Discussion

Nutrient budget data compiled by the National Rivers Authority and the Institute of Freshwater Ecology (unpublished) shows the importance of both sewage effluent and agricultural inputs as sources of P to Bassenthwaite Lake and indicates that external loading has increased in the last few decades. Analyses of nutrient load data by IFE (May *et al.*, 1995) suggested that 49% of the P entering the lake was attributable to sewage discharges, while the remainder came from diffuse sources in the catchment. Seepage from septic tanks are also likely to be important. More than 38% of the entire P load to the lake was found to be coming from a single, large sewage treatment works at Keswick. Phosphorus stripping was introduced at Keswick STW in 1995 in order to reduce nutrient loads.

A study of the geochemical profiles in the BASS1 sediment core by Morrison (1997) demonstrated that P in the sediment had increased considerably and continuously since 1900, but most markedly since c. 1970, which was attributed to an increase in output from Keswick STW and not to P mobility. The organic content in the core also increased steadily from c. 1900 and the radiometric dating results indicated a significant increase in sediment accumulation rates during the period 1900-1940 that has been more or less sustained to the present day. These data suggest that there have been changes in the lake, most likely caused by anthropogenic activity in the catchment.

It appears, however, that despite an increase in external P loads, the high flushing rate has limited the impact of eutrophication on the lake. The diatom model results suggest that there has been no increase in epilimnetic TP concentrations over the last 100 years, although diatom species shifts were observed and two taxa, not found in the 25 cm sample, were present in relatively high abundances in the upper sediment, indicating that biological changes are occurring. There is good agreement between the DI-TP value of 31 µg l^{-1} for 1995 and the current TP of the lake which ranges from 10-65 µg l^{-1} with a mean of 25 µg l^{-1}. In summary, the palaeolimnological data confirm that Bassenthwaite Lake represents a typical mesotrophic lake in a "near natural" condition, although continued monitoring of the nutrient loading to the lake and its impact on water quality is desirable. This is particularly important given that Bassenthwaite Lake supports one of only two remaining populations of Vendace (*Coregonus albula*) in the UK (Maitland & Lyle, 1991) which is known to be affected by eutrophication and particularly hypolimnetic de-oxygenation (Garner, 1996).

Figure 17 Aquatic macrophyte distribution map for Bassenthwaite Lake North

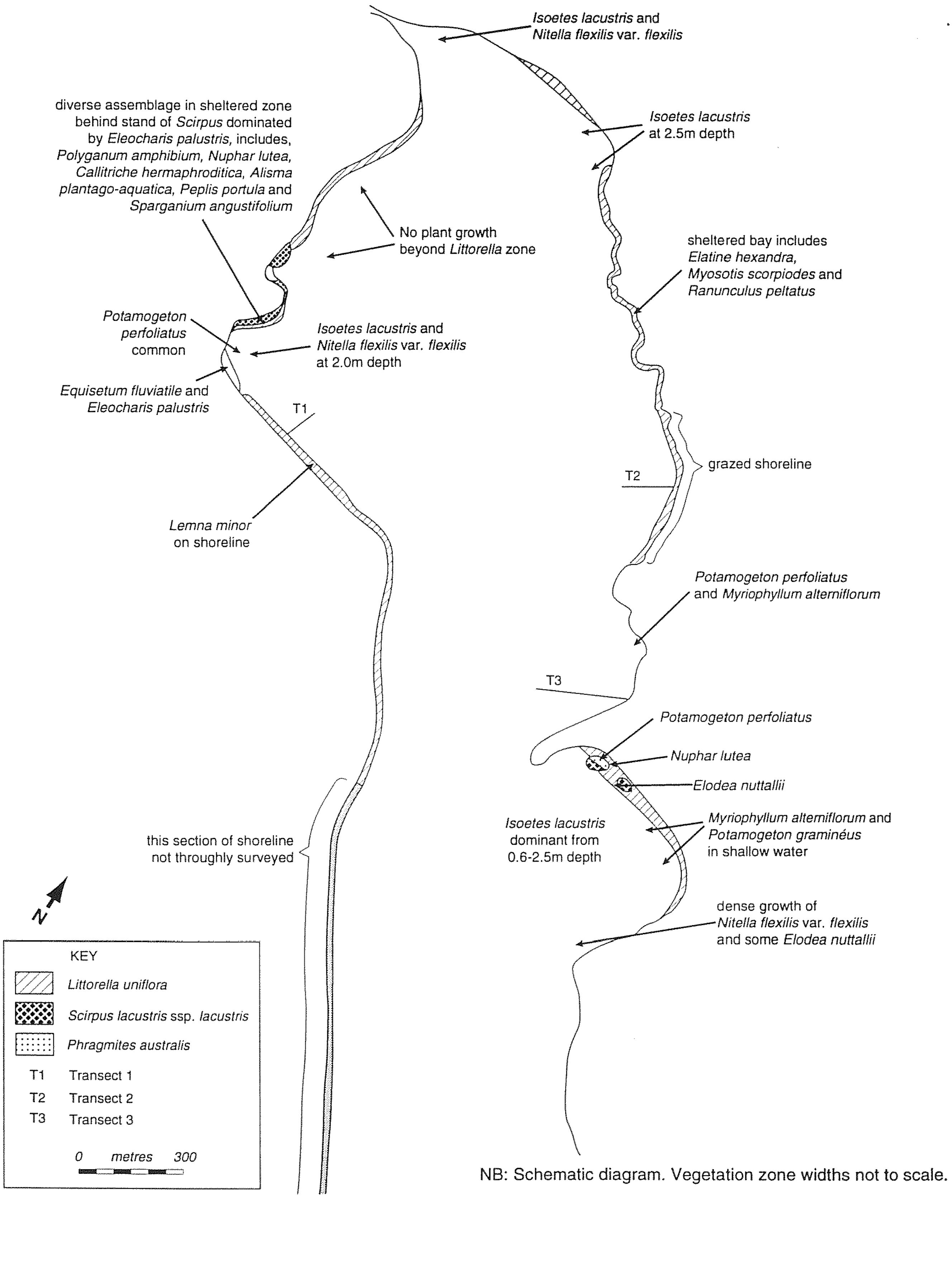
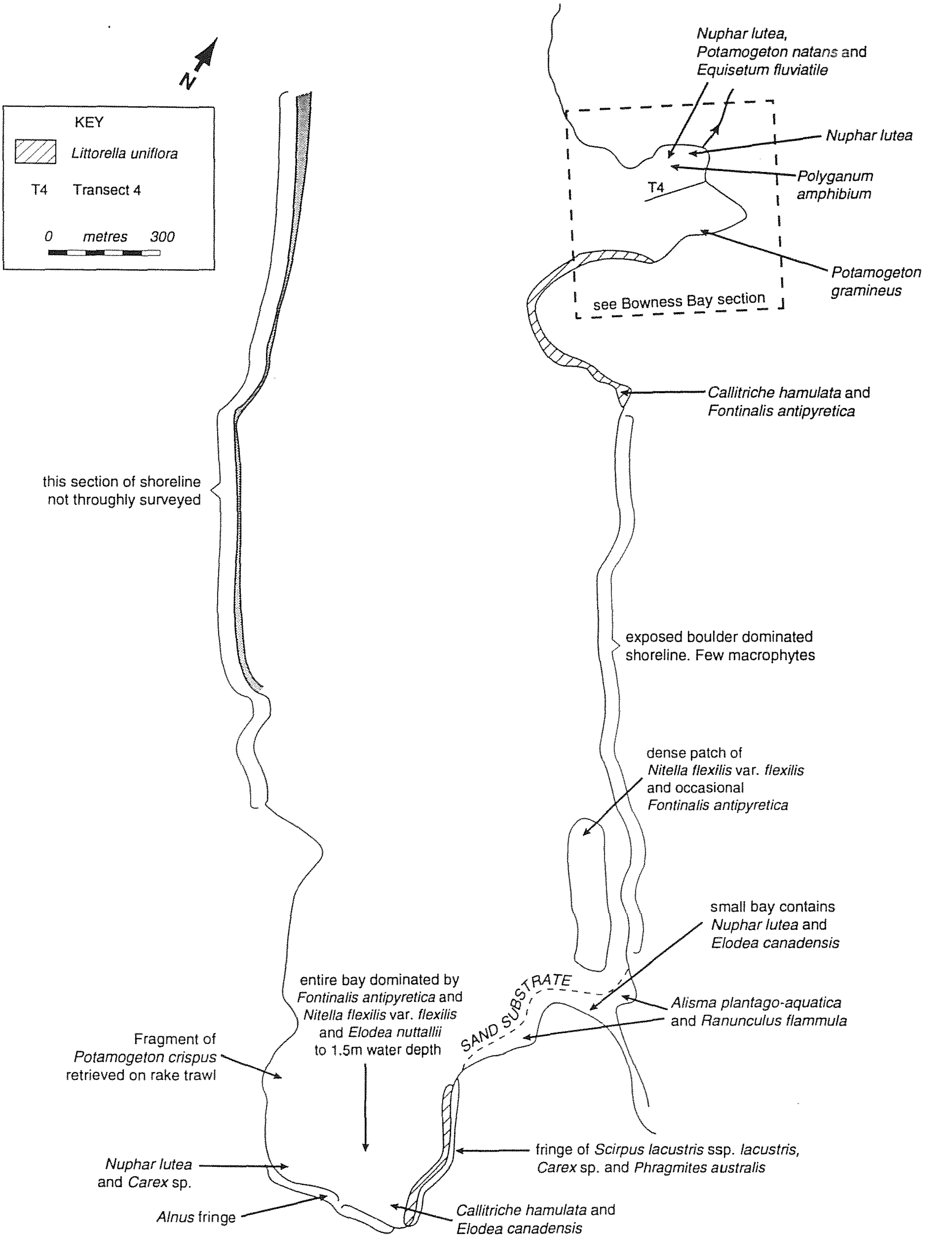
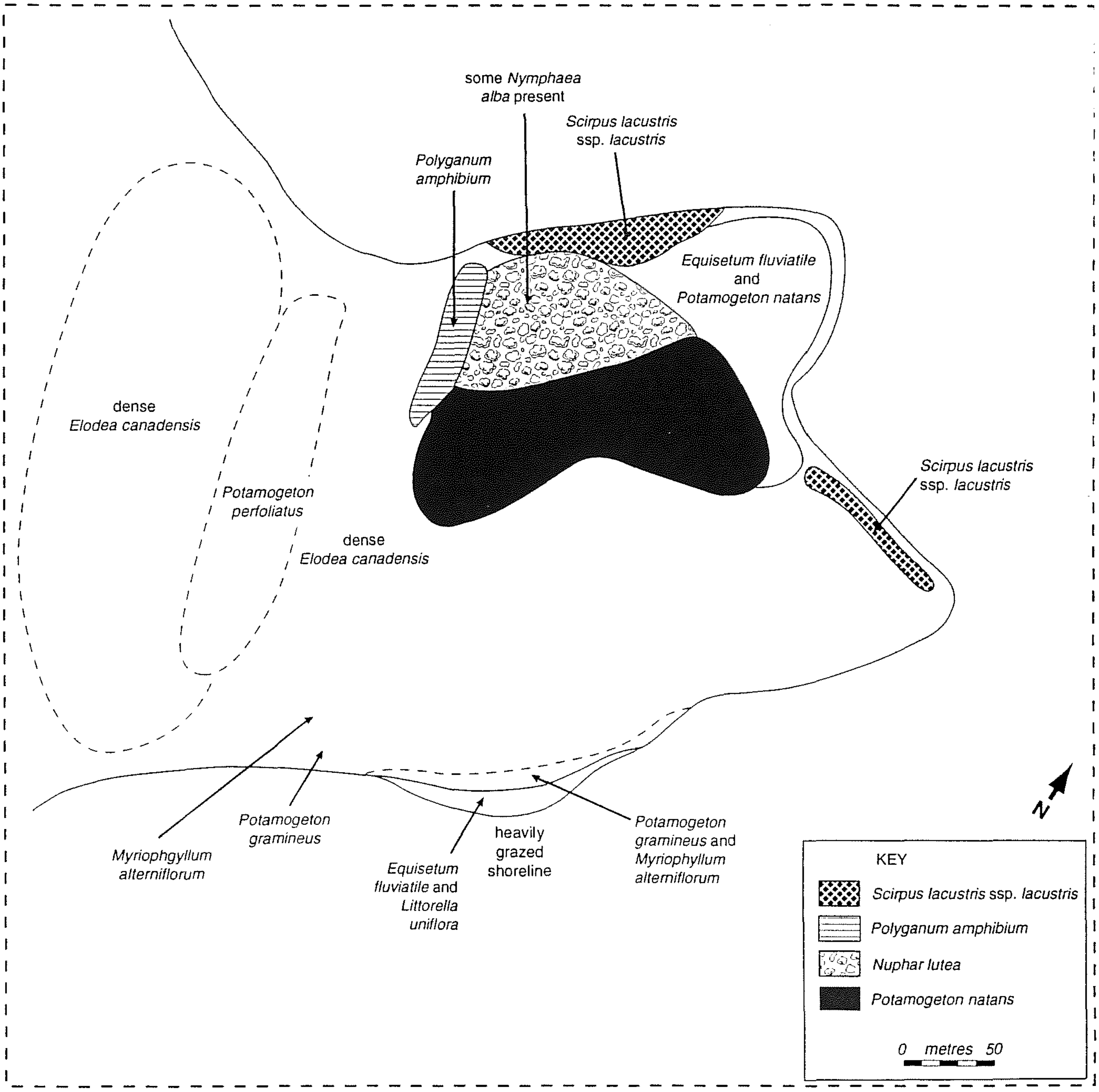


Figure 18 Aquatic macrophyte distribution map for Bassenthwaite Lake South



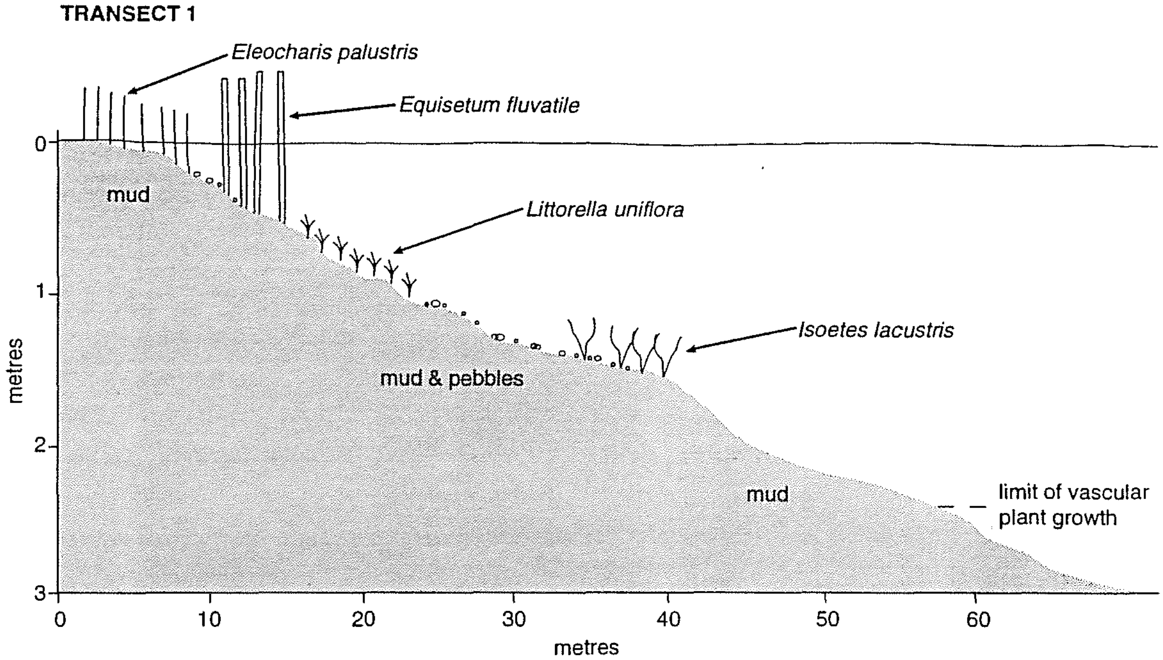
NB: Schematic diagram. Vegetation zone widths not to scale.

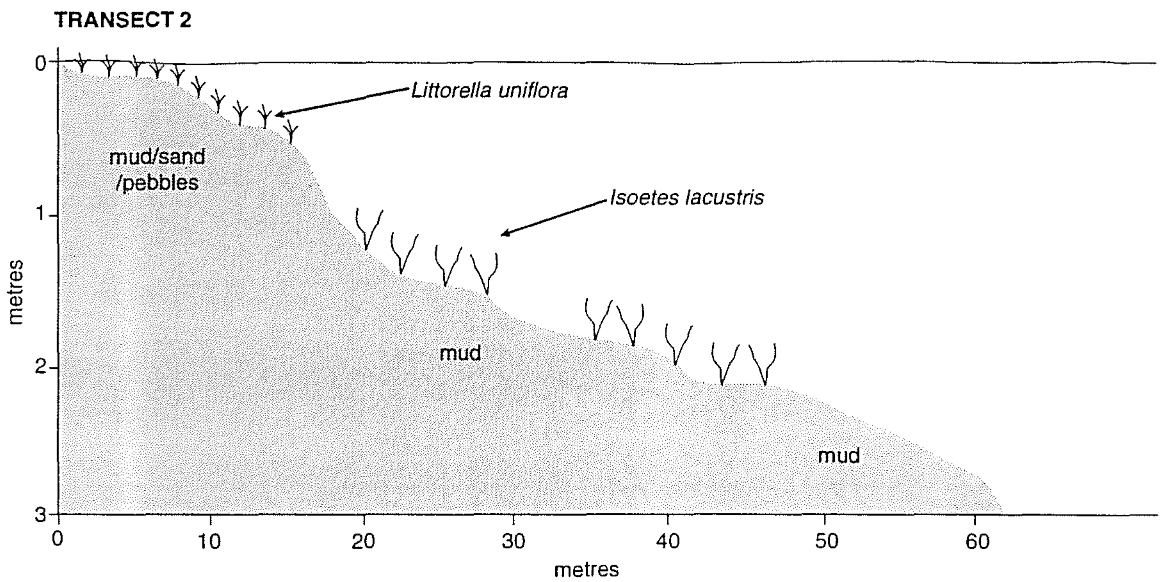
Figure 19 Aquatic macrophyte distribution map for Bowness Bay, Bassenthwaite Lake



NB: Schematic diagram. Vegetation zone widths not to scale.

Figure 20 Aquatic macrophyte transect profiles for Bassenthwaite Lake





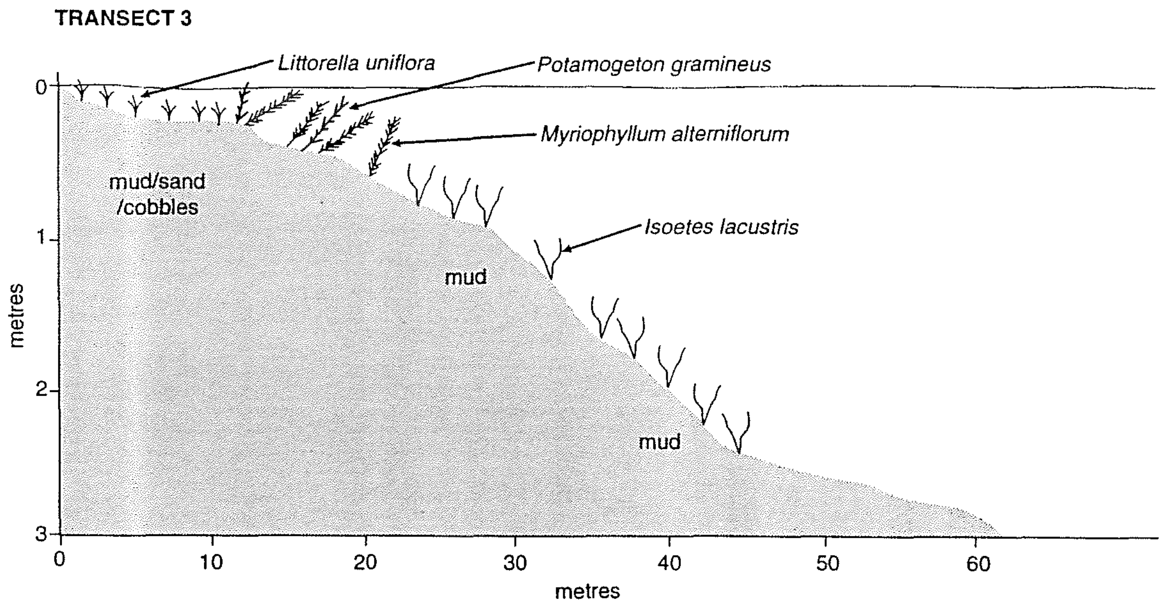


Figure 21 Lithostratigraphic data for Bassenthwaite Lake

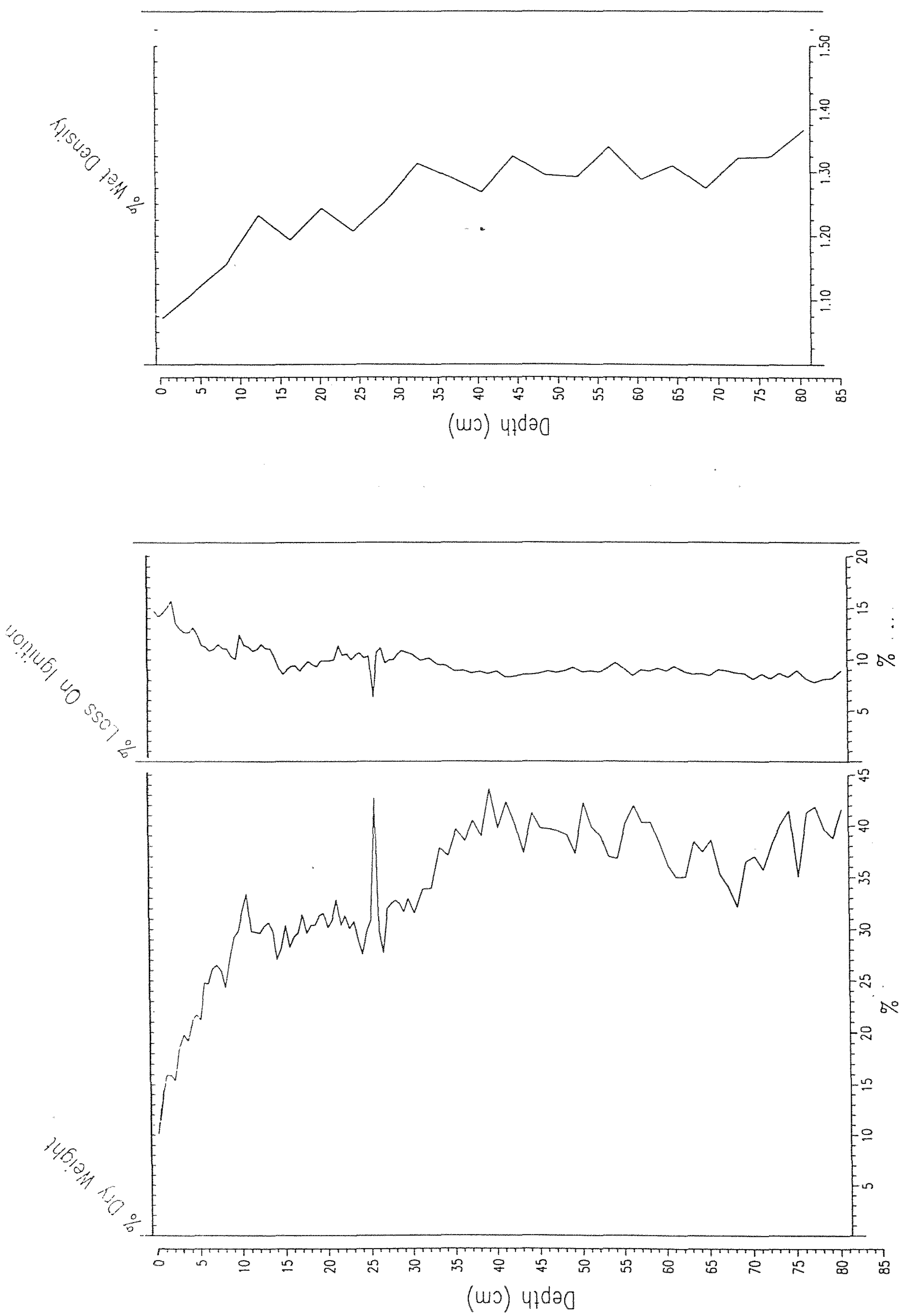


Figure 22 ^{210}Pb activity versus Depth - Bassenthwaite Lake

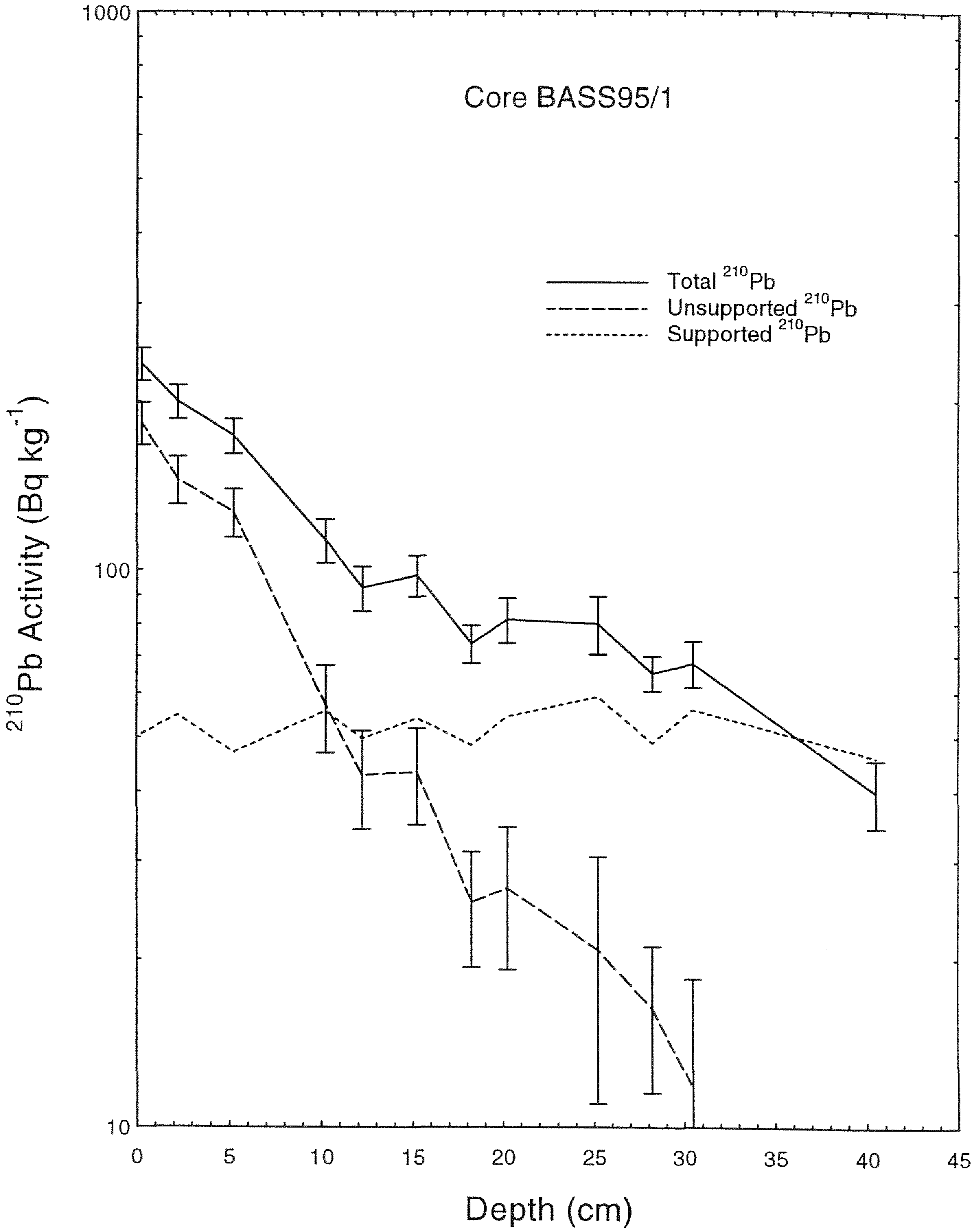


Figure 23 ^{137}Cs and ^{241}Am Activity versus Depth - Bassenthwaite Lake

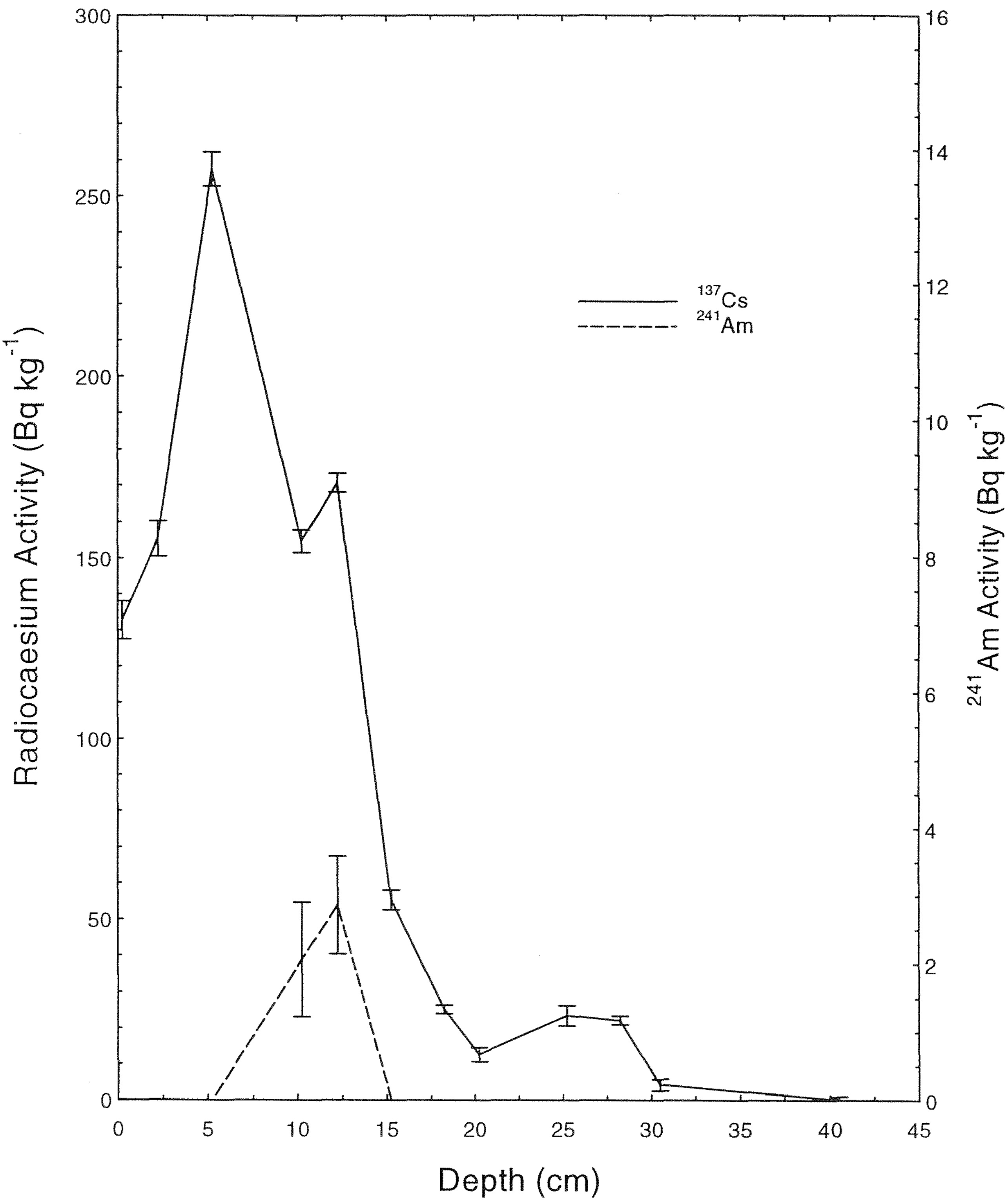


Figure 24 Depth versus Age - Bassenthwaite Lake

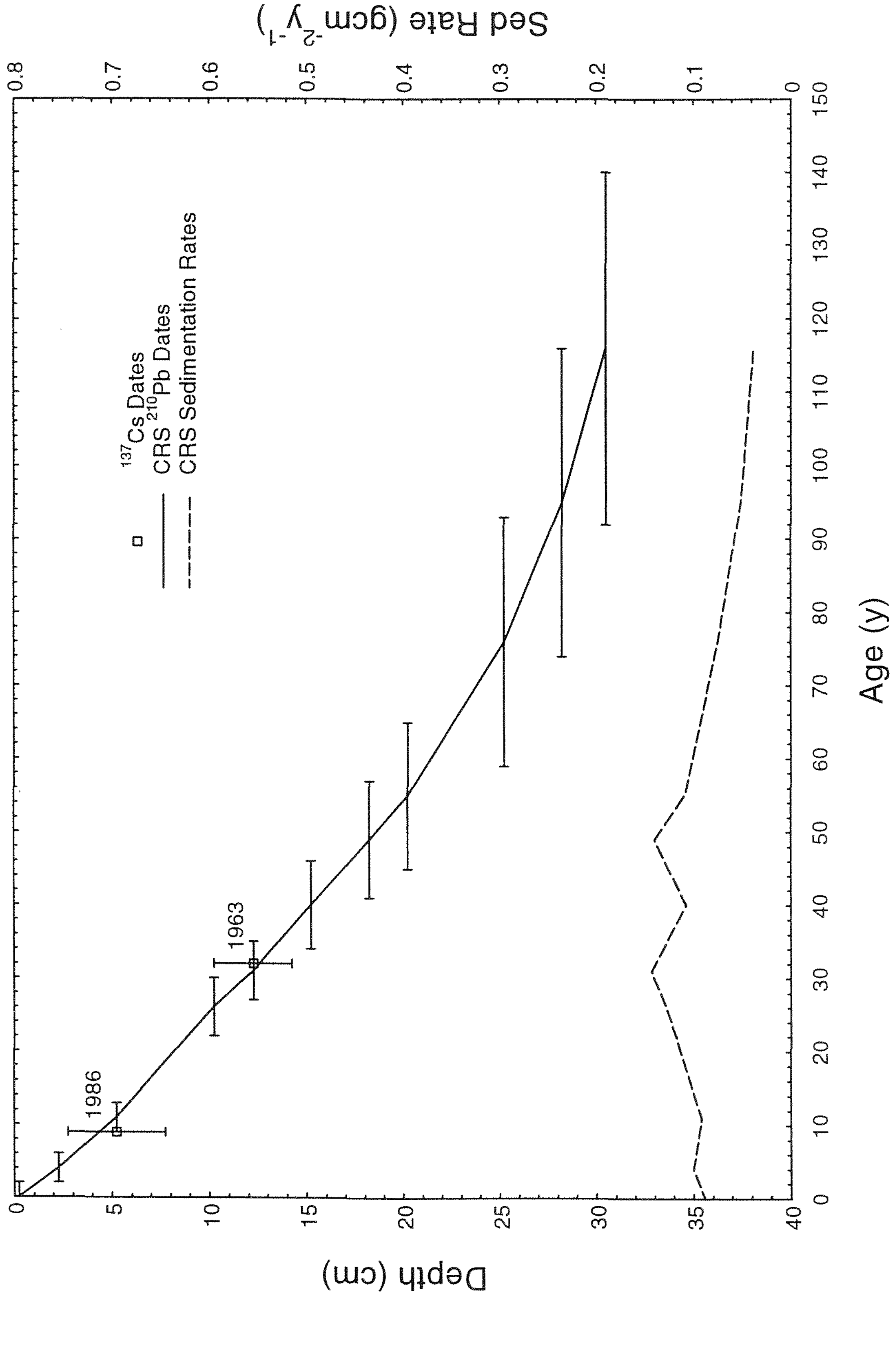
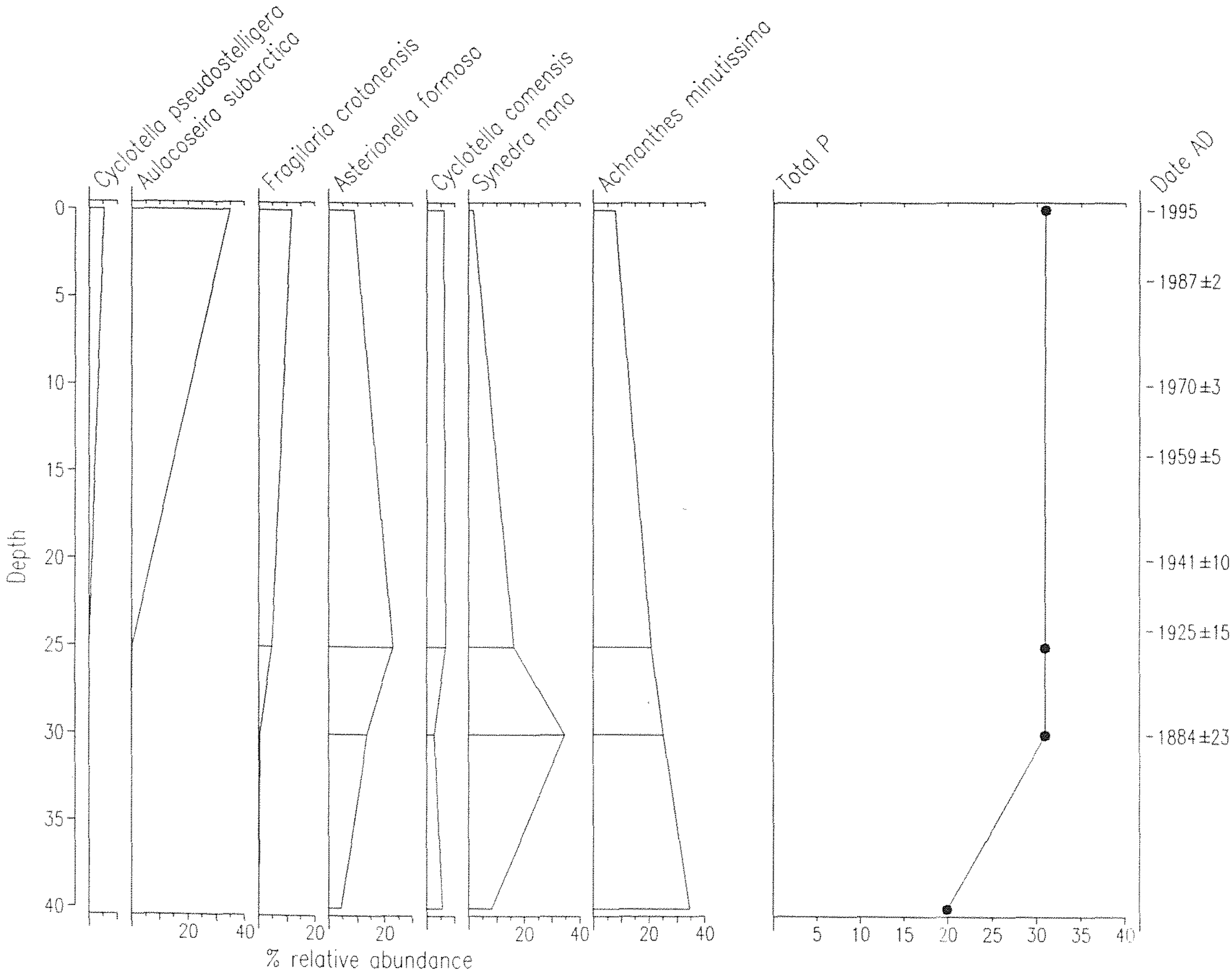
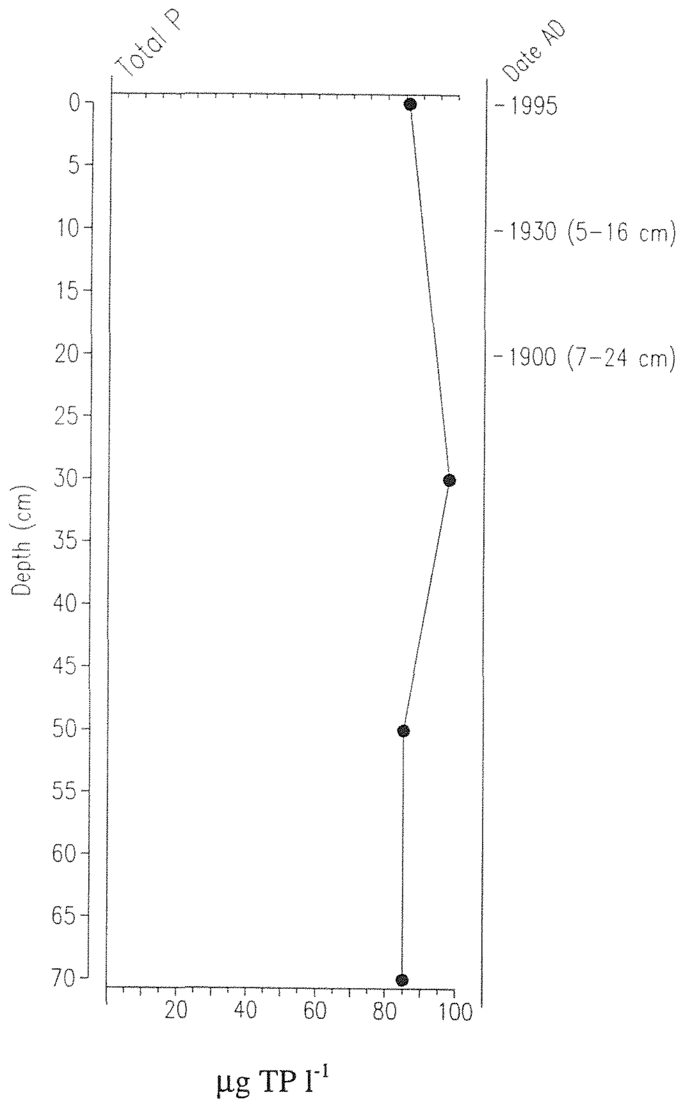


Figure 25 Summary diatom diagram and reconstruction for Bassenthwaite Lake



Greenlee Lough

- The DI-TP results indicate that Greenlee Lough has not experienced any major changes in nutrient concentrations and represents a mesotrophic lake in a near-natural condition.
- However, the data should be interpreted with some caution owing to a high percentage of poor trophic-indicator diatom taxa in the core and the poor temporal resolution of the sediment record.
- A number of diatom taxa associated with more eutrophic waters, *Stephanodiscus hantzschii, Synedra acus* and *Asterionella formosa*, were present in the surface sample (though be it in very small abundances of < 2%).



8. Greenlee Lough, Northumbria (NY 774 698)

8.1 Site Description and Water Chemistry

- Greenlee Lough is situated due north of Bardon Mill and Hadrian's Wall and is part of the Roman Wall Loughs SSSI within the Northumberland National Park.
- The lough lies on lower carboniferous sedimentray rocks consisting of limestones, shales, sandstones and thin coals
- Altitude 229 m
- Maximum depth 1.8 m
- Area 0.45 km^2
- Land use is mostly rough grazing with some improved pasture and afforestation

pH	units	7.4
conductivity	$\mu S\ cm^{-1}$	154
alkalinity	mg $CaCO_3$	45
calcium	mg l^{-1}	18.3
total nitrogen	mg l^{-1}	0.8
total phosphorus	$\mu g\ l^{-1}$	16

nb. These readings were taken from the lake outflow.

8.2 Macrophyte Survey

Date of visit 11-6-95.
An aquatic macrophyte distribution map for Greenlee Lough is provided in Figure 26. The site is shallow (maximum depth 1.8m) and, despite poor water clarity (secchi disc depth 1.2 m), submerged macrophytes cover much of the lake bed.

Much of the shoreline around the western half of the site is fringed by beds of emergent macrophytes. *Carex rostrata* is dominant in many sheltered bays, often flanked on the open water side by stands of *Equisetum fluviatile*. At the outflow end these communities grade towards the shore into *Phragmites australis* swamp. Several stands of *Eleocharis palustris* also occur in shallow water habitats.

A wide but relatively sheltered bay on the north-east shoreline contains an association of *C.rostrata*, *Hippuris vulgaris* (in very shallow water), *Alisma plantago-aquatica*, *Sparganium erectum* and *Callitriche* sp.. The bay at the far east end of the lake is very shallow (0.1-0.2 m) with a sand substrate which supports *Juncus bulbosus* var. *fluitans*, *Nitella flexilis* var. *flexilis* and *Potamogeton gramineus.*

The southern shore of the east end of the lake is characterised by an exposed, boulder dominated substrate and offers no suitable habitat for macrophyte colonisation. However *Fontinalis antipyretica* and *Chara* sp. thrive in deeper water off-shore.

Two, partly artificial, deeply cut bays contain a rich species assemblage, highlighting the importance of physical habitat for species diversity at this generally wind exposed site. Within this area the shoreline emergent community includes *C.rostrata*, *Typha angustifolia*, *A. plantago-aquatica*, *Eleocharis palustris*, *S. erectum*, *S. emersum*, *Menyanthes trifoliata* and *E. fluviatile* while *Potamogeton crispus*, *P. gramineus*, *P.natans*, *P.berchtoldii, Elodea canadensis and J.bulbosus* var. *fluitans* are all found in shallow water.

The open water habitat throughout much of the lake is dominated by thick beds of *Chara* sp. and *F. antipyretica*, with *N. flexilis* var. *flexilis* locally abundant towards the outflow end.

Greenlee Lough was surveyed for aquatic macrophytes by Palmer in 1982 for the Nature Conservancy Council. The species list from that survey is very similar to the one given in Table 7 which suggests that environmental conditions have not altered much in the last decade. However, certain taxa found in 1982 were not detected during our survey. These include *Myriophyllum alterniflorum*, *Utricularia australis/vulgaris* and *Potamogeton perfoliatus.* The survey team were hampered by poor weather and particularly strong winds which made working from the inflatable survey boat in open water problematic. It is therefore possible that these three species were overlooked. It is interesting to note that the relatively newly introduced species *E. canadensis*, recorded as "dominant over much of the lough" in the 1982 survey has been found in very few locations during our survey. However, this species dies down in winter and regrowing from underground stems in spring, and it is likely to become more abundant later in the summer.

Greenlee Lough is typed, according to Palmer (1992), as type 5a, a mesotrophic category.

Table 7 Species list and DAFOR abundance rating for Greenlee Lough

	abundance	notes
Submerged taxa		
Chara sp.	A	locally dominant
Nitella flexilis var. *flexilis*	O	locally abundant towards outflow
Fontinalis antipyretica	A	widespread in deeper water
Callitriche sp.	O	widespread
Elodea canadensis	R	in south
Littorella uniflora	O	locally abundant in north-east
Potamogeton alpinus	R	in north-east
P. berchtoldii	O	in west
P. crispus	O	locally abundant around pier area
P. gramineus	F	locally abundant in shallow water
P. pectinatus	R	in inflows
P. pusillus	R	in west
Juncus bulbosus var. *fluitans*	O	mostly in north-east bay
Floating leaved taxa		
Potamogeton natans	O	in east
P. polygonifolius	R	in north-west
Sparganium emersum	O	locally abundant in east
Emergent taxa		
Hippuris vulgaris	R	in north-east
Phragmites australis	F	locally dominant in west
Phalaris arundinacea	O	
Typha latifolia	O	
Typha angustifolia	R	
Carex rostrata	A	locally dominant in sheltered bays
Carex paniculata	R	
Sparganium erectum	O	mainly in sheltered bays in south
Eleocharis palustris	A	
Alisma plantago-aquatica	F	
Caltha palustris	R	
Eleocharis palustris	A	
Mentha aquatica	O	
Ranunculus flammula	F	
Potentilla palustris	O	
Scirpus lacustris ssp. lacustris	R	locally abundant in west
Iris pseudacorus	O	in west

8.3 Lithostratigraphy

A 73 cm sediment core (GREE1) was taken in a water depth of 1.8 m using a Mackereth corer on 11-6-95. The lower part of the sediment core (> 20 cm) was a black (10YR 2/1) silt with some fine sand and plant remains (Ld2 Ag2 Lso+ Ga+ Dh+).There was a slight colour change at 20 cm to a very dark grey (10YR 3/1) and the silt content of the sediment decreased (Ld3 Ag1 Ga++ Lso+). There was a clear layer of more minerogenic material at c. 4-8 cms. The uppermost 3 cms were a very dark brown (10YR 2/2), more organic lake mud with some silt (Ld3 Ag1 Lso+ Dg+ Ga+).

The %dw and %loi profiles (Figure 27) fluctuated considerably throughout the core. %dw generally varied between 20 and 40%, and %loi between 10 and 30%. This variability suggests that there have been periods of frequent minerogenic inwash and probably also a fair degree of sediment mixing (given that the site is wind stressed and shallow). There was a distinctive layer of much denser material between 4 and 8 cms, where %dw rose to 65% and %loi fell to 2%. Even the uppermost sediments were inorganic with a %loi value of only 5%. The wd data followed the same trend, values fluctuating between 1.10-1.25 g cm^{-3} throughout the core, except for a clear layer at c. 5 cm where values reached 1.40 g cm^{-3}.

8.4 Radiometric Dating

Unsupported ^{210}Pb activities in this core were negligible except in the top 2-3 cm and were insufficient to allow dating by the ^{210}Pb method. The ^{210}Pb inventory of the top sections was just 438 Bq kg-1, less than 20% of the fallout record. The reasons for this anomaly are almost certainly associated with the layer of unusually dense sediment at c.6 cm, though there was no recovery in ^{210}Pb values in the apparently normal deeper section at 12 cm (Figure 28).

The ^{137}Cs profile had a well defined maximum at 2.25 cm, though it was not clear whether this feature records the 1986 Chernobyl accident or the 1963 weapons fallout peak (Figure 29). Unusually, the ^{137}Cs inventory of the core was significantly higher than the ^{210}Pb inventory.

In view of these uncertainties it would be unsafe to make any estimate of accumulation rates beyond suggesting that they are in the region of 0.1-0.25 cm y^{-1}. This would put 1930 at 5-16 cm and 1900 at 7-24 cm.

8.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in four levels of the sediment core were calculated and Figure 30 illustrates the results for the major taxa. Diatom preservation was good to 30 cm downcore but was poor in the more inorganic, lower core section, where mineral matter also caused interference problems. A total of 80 taxa was observed, 64 of which were present in the calibration set. All of the common taxa were well represented in the calibration set, with greater than 88% of the fossil assemblage being used in the calibration procedure.

Figure 30 illustrates that there have been slight changes in the diatom species composition in terms of relative percentages over the period represented by the 70 cm core. It is not possible to estimate the time represented by the core because of the problematic radiometric dates, other than to say that 1930 lies between 5 and 16 cm, and 1900 between 7 and 24 cm. There have been no clear species replacements and the assemblages have been dominated by the same major, non-planktonic taxa throughout the core: the benthic *Fragilaria* spp., *Achnanthes lanceolata, Amphora pediculus* and *Navicula cocconeiformis*, species commonly found attached to either the sediments or macrophyte surfaces of shallow, alkaline waters. The only changes were that *Fragilaria construens* var. *venter* and *Navicula cocconeiformis* accounted for greater relative abundances in the lower samples (50 and 70 cm) than in the upper ones (0 and 30cm). A further observation was that *Achnanthes minutissima*, an epiphyte commonly found in freshwaters, increased in importance in the upper sediments and a number of diatom taxa associated with more eutrophic waters, *Stephanodiscus hantzschii, Synedra acus* and *Asterionella formosa*, were present in the surface sample (though be it in very small abundances of < 2%).

8.6 Total Phosphorus Reconstruction

The TP reconstruction shows that concentrations have remained largely unchanged throughout the period represented by the core. The model indicates that the lake has been eutrophic for the whole of this period with values of around 85-90 μg TP l^{-1}.

8.7 Discussion

There are few nutrient chemistry data available for Greenlee Lough and so it is not possible to directly compare the diatom-inferred concentrations with measured values. The site has been classed as mesotrophic based on invertebrate and macrophyte survey data and on chemical data collected by the Institute of Terrestrial Ecology and reported in Palmer (1982), where summer TP concentrations were in the range 19-26 μg TP l^{-1}. It is usual that TP concentrations are at their maximum in winter and therefore the annual mean TP values are probably greater than the concentrations measured in the summer survey. Water samples collected by the EA from the lake outflow, however, also give low readings of TP ranging from 10-18 μg TP l^{-1} over the period April 1995 to September 1996.

It does seem, therefore, that the diatom model over-estimates the TP values for Greenlee Lough. This is most likely because of the lack of planktonic forms in the fossil assemblages. The dominance of non-planktonic *Fragilaria* spp. has been observed in other shallow, alkaline British waters (Bennion, 1994; 1995). They are known to be poor indicators of lake trophic status because of their wide ecological tolerances and associations with substrates (plants, stones, mineral particles) rather than directly with epilimnetic water chemistry. Furthermore, the radiometric dating results indicate that the sediment record may be disturbed and therefore the temporal resolution is poor. These problems should be considered when interpreting the model results for Greenlee Lough.

In summary, Greenlee Lough does not appear to have experienced any major changes in water quality as inferred by the diatom model, and probably represents a mesotrophic lake in a near natural condition. The appearance of a number of taxa indicative of more productive waters in the surface sample should be noted, however, as this suggests that subtle ecological changes are occurring in the lake, possibly signalling the start of enrichment. Similar findings were also observed by Lewis (1996) in her study of a sediment core from the lough. Lewis (1996) also performed a detailed palaeolimnological study of the neighbouring Crag Lough which revealed that the diatom community of the lough had been stable until the 1970s. There were significant changes over the last few decades with increases in the taxa indicative of high P concentrations, such as *Stephanodiscus parvus* and *S. hantzschii*, signalling a process of recent eutrophication.

Figure 26 Aquatic macrophyte distribution map for Greenlee Lough

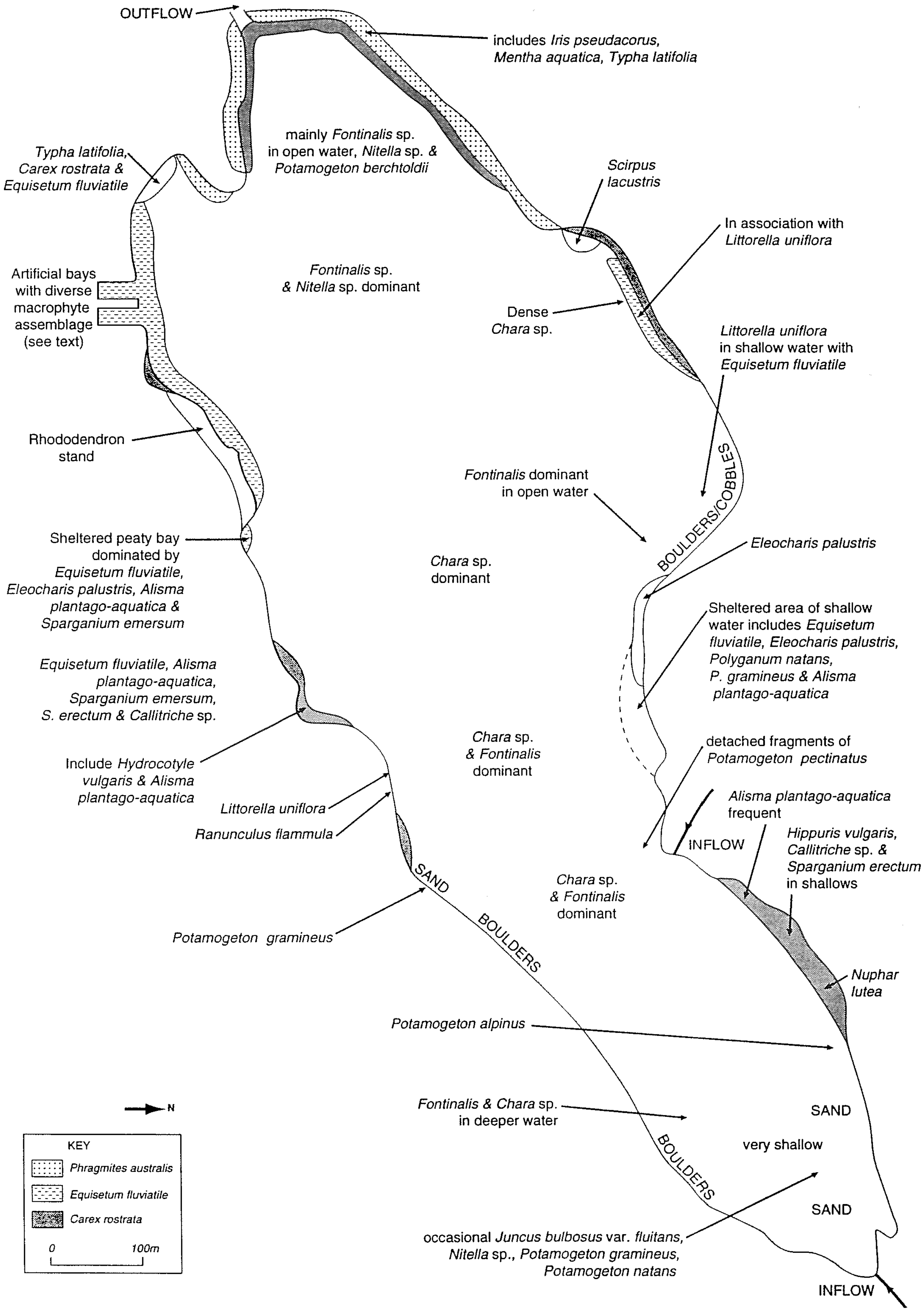


Figure 27 Lithostratigraphic data for Greenlee Lough

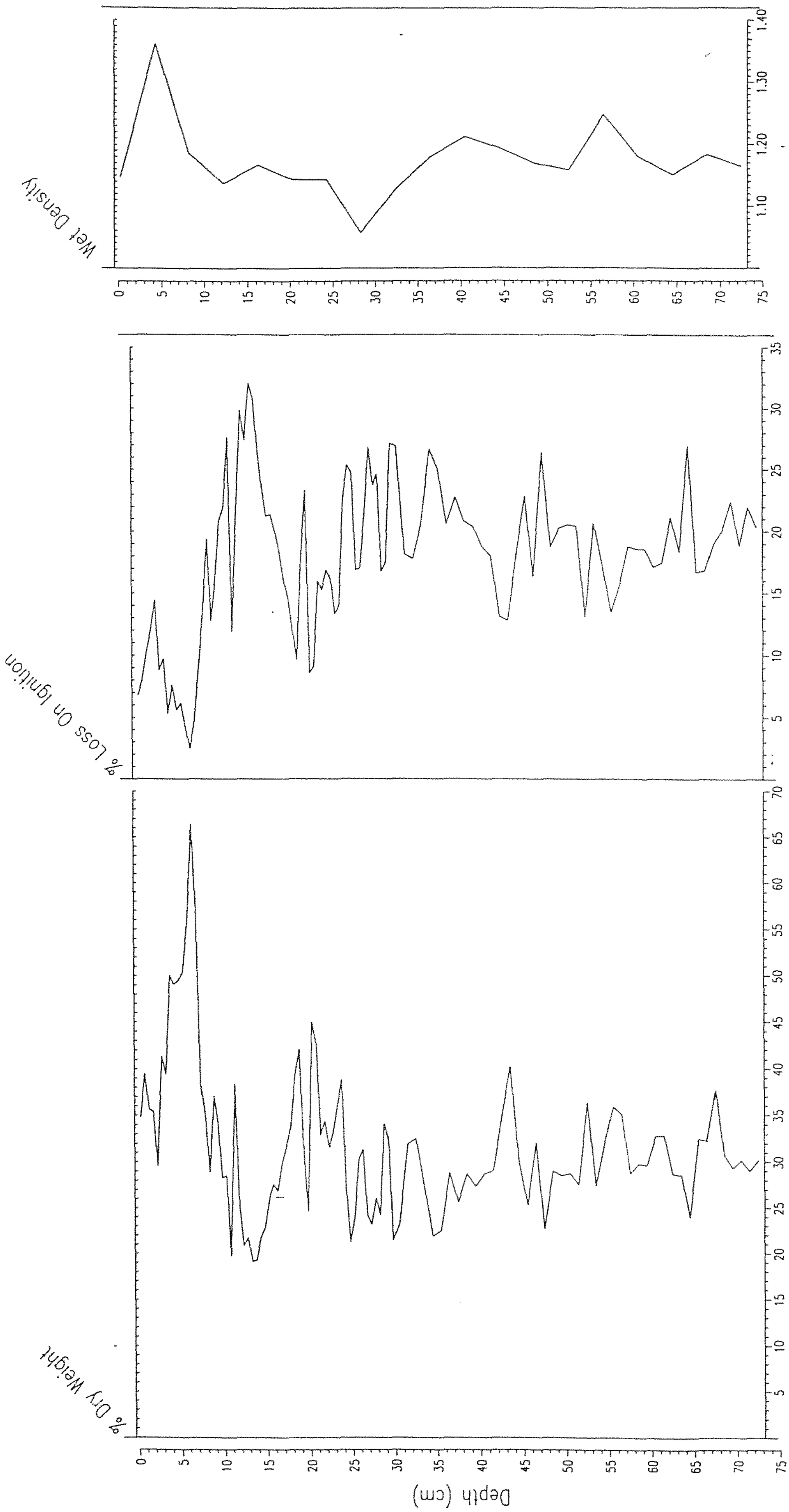


Figure 28 ^{210}Pb Activity versus Depth - Greenlee Lough

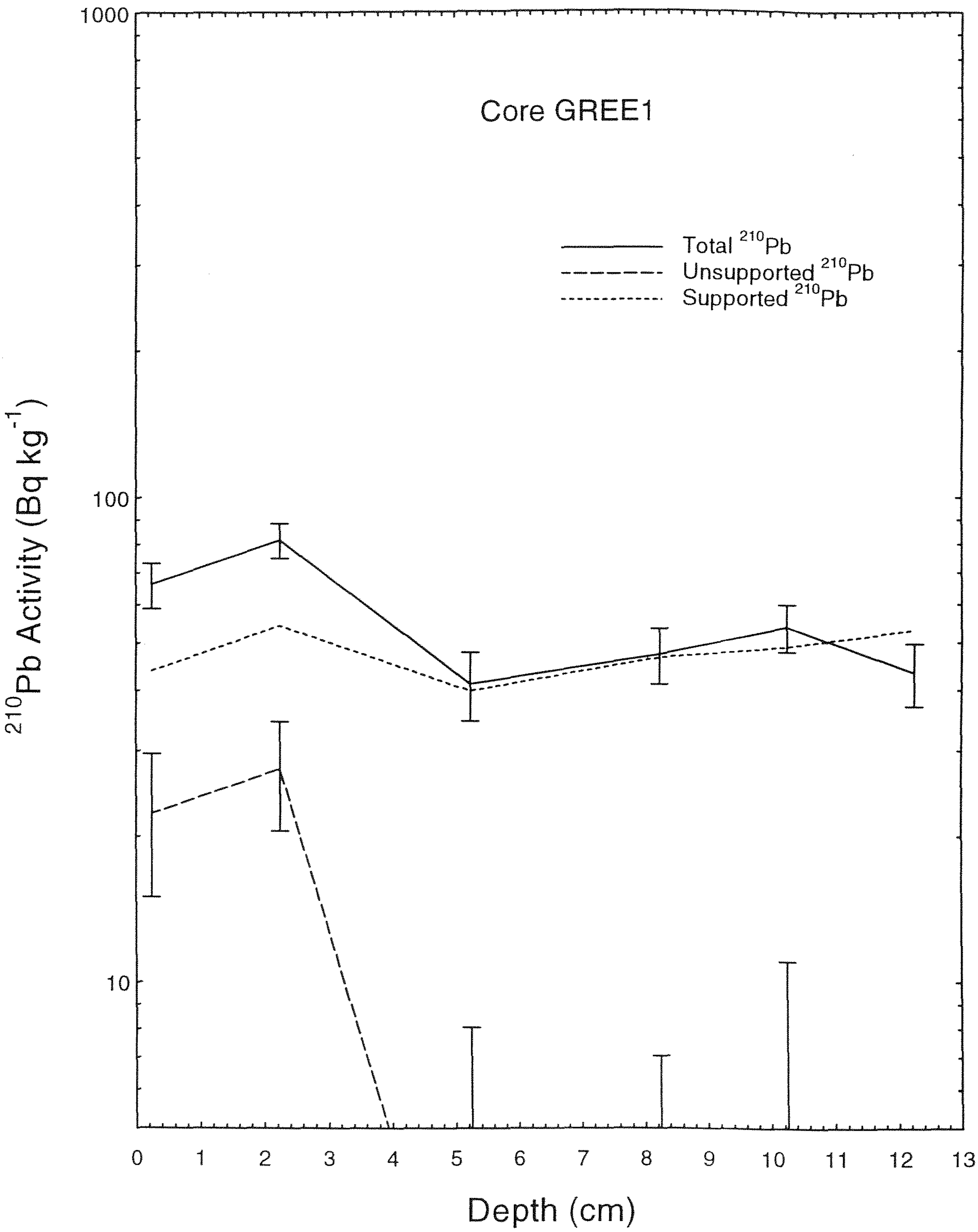


Figure 29 ^{137}Cs Activity versus Depth - Greenlee Lough

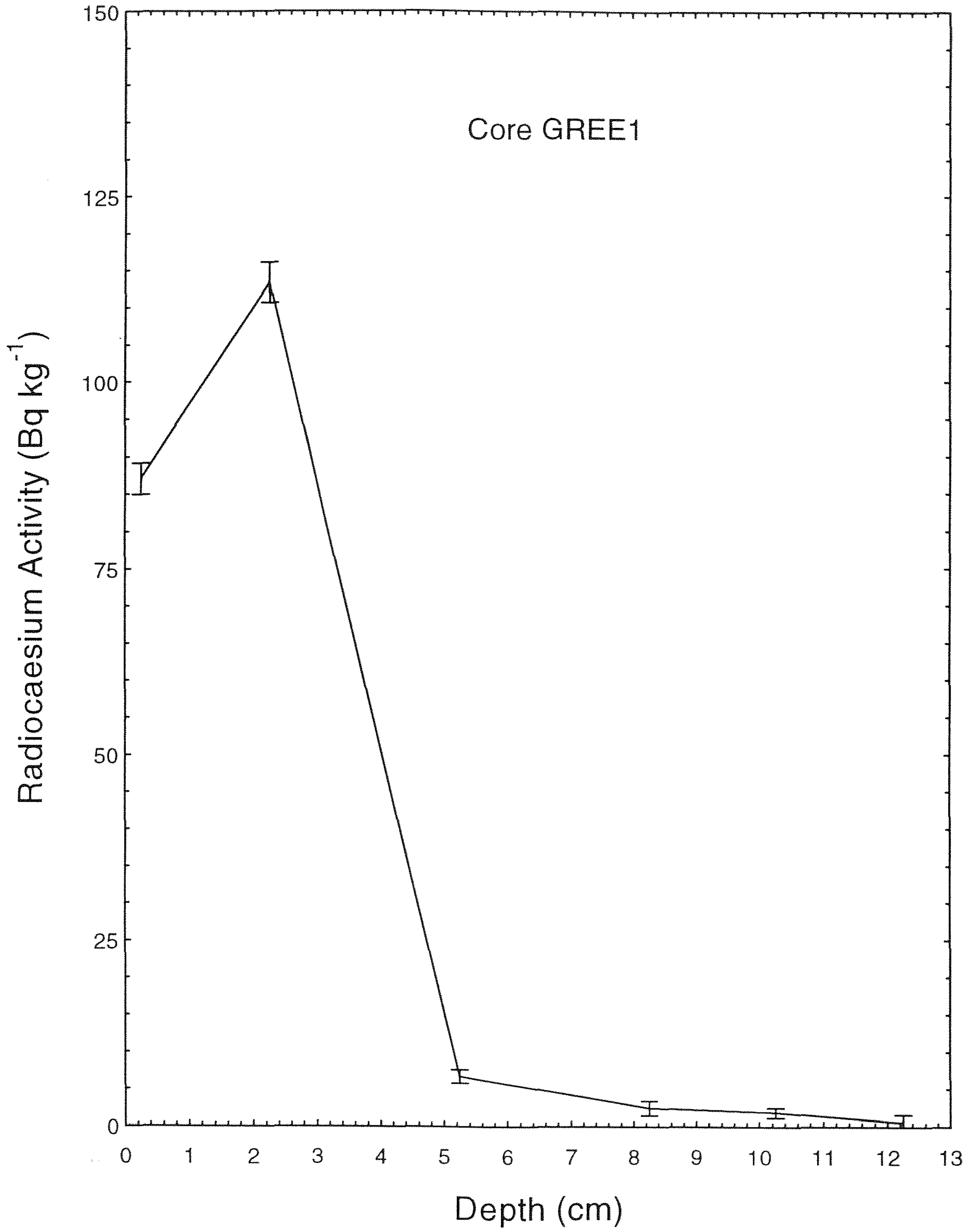
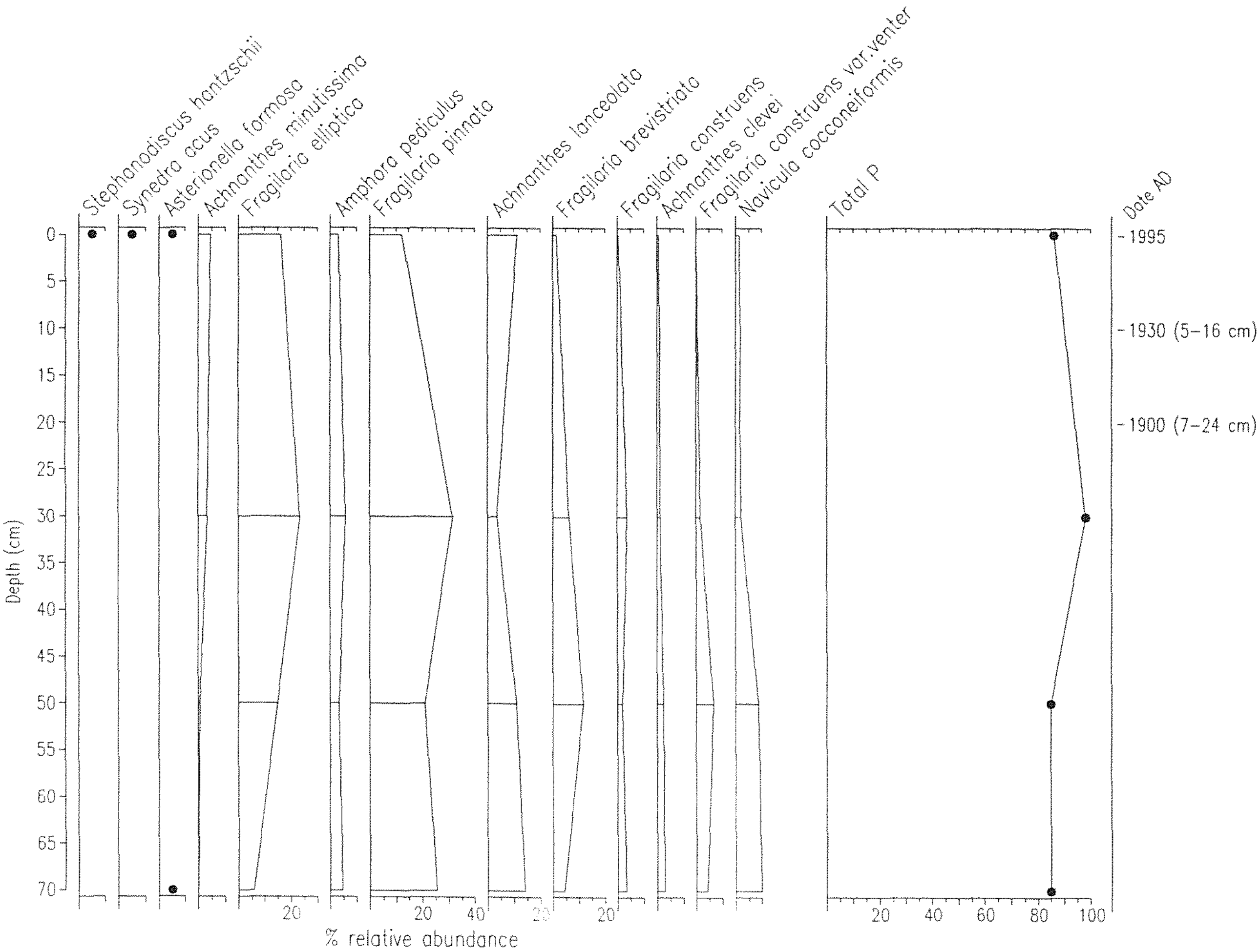


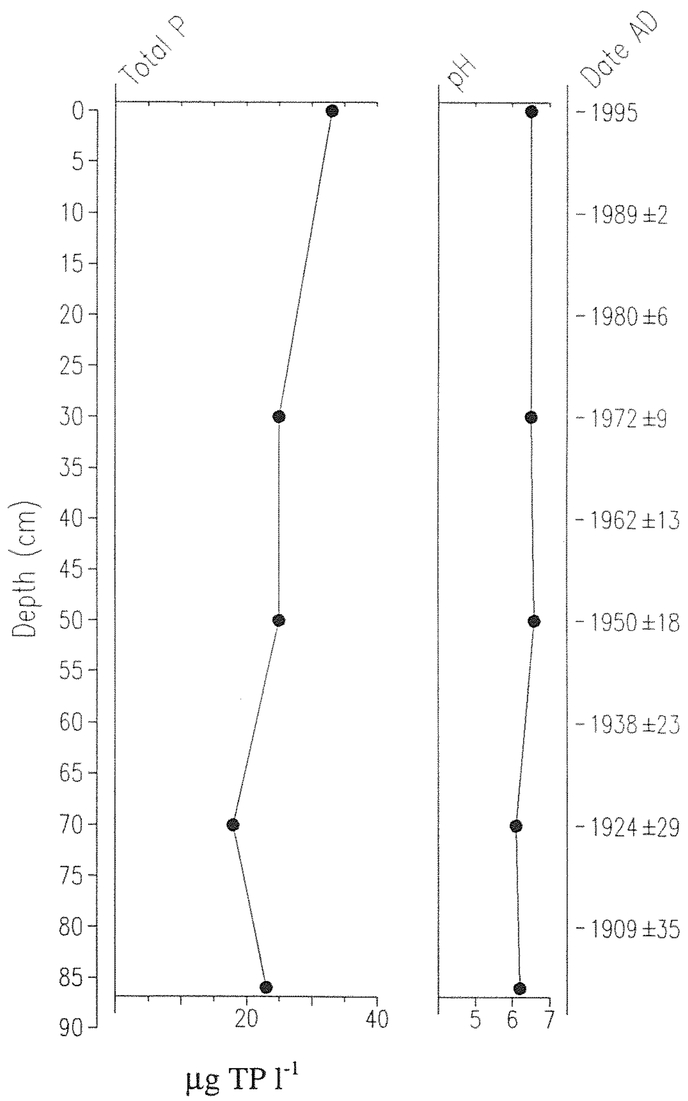
Figure 30 Summary diatom diagram and reconstruction for Greenlee Lough



(Circle in graphs for the first 3 taxa indicates presence of < 2%)

Hatchet Pond

- **The DI-TP results indicate that Hatchet Pond has been a mesotrophic lake since at least 1900 with evidence of slight enrichment in the last two decades.**
- **The DI-pH model indicates that the lake has become less acid since c. 1950 but the change is very slight and may not be significant.**



9. Hatchet Pond, Hampshire (SU 367 016)

9.1 Site Description and Water Chemistry

- A small (800 m by 200 m), shallow pond situated in the New Forest
- Altitude 35 m
- Maximum water depth 3.5 m
- Patchy reed bed areas
- Surrounded by heathland and scrub with some mixed woodland in the catchment
- Lying on sandy soils and gravels

pH	units	7.2
conductivity	$\mu S\ cm^{-1}$	127
nitrate	$mg\ l^{-1}$	0.17
orthophosphate	$\mu g\ l^{-1}$	11

nb. These readings were taken from Hatchet Stream

9.2 Macrophyte Survey

Date of visit 26-7-96.
An aquatic macrophyte species distribution map is presented in Figure 31. The aquatic macroflora is generally indicative of a mesotrophic water body. Despite being shallow, with a maximum depth of less than 1.5 m, aquatic macrophytes are generally restricted to a depth of no more than 0.8 m, probably as a result of low water transparency (secchi disc depth 0.5 m). The littoral substrate is dominated by mud and fine sand. Two main aquatic macrophyte community types are apparent.

The main section of the lake, comprising the length that lies approximately east - west is characterised by a submerged littoral association of *Littorella uniflora*, *Myriophyllum alterniflorum*, *Elodea canadensis* and the charophyte *Nitella translucens*, the latter two species being more abundant in deeper water between 0.5 - 0.8 m. With the exception of a detached shoot of *Potamogeton crispus* found amongst strand-line material, no other submerged species were detected. In very shallow areas in the eastern part of the pond, *Ludwigia palustris* is locally abundant and often growing in association with *Hypericum elodes*. At the far eastern end *Eleocharis acicularis* and *Elatine hexandra* are locally dominant on the shoreline.

The most diverse aquatic flora is found in the north end of the pond which contains a number of stands of emergent taxa. *Menyanthes trifoliata* is most dominant here, although *Typha latifolia* and *Equisetum fluviatile* are also locally abundant, and other species present include *Luwigia palustris*, *Potamogeton natans*, *Sagittaria sagittifolia*, *Sparganium angustifolium* and a pink leaved hybrid of *Nymphaea lutea* (probably of horticultural origin - see Preston and Croft, 1997). *Scirpus fluitans*, *Juncus bulbosus* var. *fluitans* and *Elodea canadensis* are found here in submerged habitats.

Few previous survey records were available for Hatchet Pond, but the site is clearly a biologically diverse habitat for both emergent and submerged species despite the current poor water transparency and there is little indication of it having deteriorated over recent times. In the Nature Conservation Review, Ratcliffe (1977) refers to the occurence of *Pilularia globulifera*, a species which was not found during this survey. There is some evidence that *P. globulifera* has been lost from many sites in the UK over the last century as a result of successional changes brought about by grazing, but it is thought that its current UK distribution is more or less stable (Preston and Croft, 1997). It is therefore unwise to suggest that this species may have been lost from Hatchet Pond on the basis of one survey. It is likely that the turbidity is largely due to resuspension of sediment from the lake bed, an inevitable consequence at such a shallow site with relatively large surface area.

Applying the species list in Table 8, Hatchet Pond is classified as Type 2 (Palmer, 1992), an oligotrophic category.

Table 8 Species list and DAFOR abundance rating for Hatchet Pond

	abundance	notes
Submerged taxa		
Elodea canadensis	F	throughout in open water
Elodea nutallii ?	R	identification uncertain
Littorella uniflora	A	the dominant littoral macrophyte in the w arm
Juncus bulbosus var. *fluitans*	R	n-e end
Myriophyllum alterniflorum	F	throughout in open water
Nitella translucens	F	throughout in open water
Potamogeton crispus	R	in strand-line debris only
Scirpus fluitans	R	n-e end
Floating leaved taxa		
Nymphaea alba hybrid ?	R	n-e end
Potamogeton polygonifolius	O	w end
Emergent taxa		
Elatine hexandra	O	w end
Eleocharis acicularis	O	w end
Equisetum fluviatile	O	small stands in n-e end
Hypericum elodes	O	throughout
Juncus articulatus	O	n-e end
Juncus bulbosus var. *fluitans*	O	n-e end
Ludwigia palustris	F	locally abundant in shoreline habitats throughout
Menyanthes trifoliata	F	locally abundant in n-e end
Ranunculus flammula	R	throughout
Sagittaria sagittifolia	R	n-e end
Sparganium emersum	R	n-e end

9.3 Lithostratigraphy

An 88 cm sediment core (HATC1) was taken from the deepest part of the lake in a water depth of 3.3 m using a Mackereth corer on 18-1-95. The lower sediments (> 50 cm) were a very dark grey (10YR 3/1), lake mud with a high clay content and plant remains, and diatoms were present to the core base (Ld2 As1 Lso1 Dg++). There was a colour change at c.50 cm to a very dark grey (10YR 2/2) and the clay content of the sediment decreased. Diatoms and detritus were more abundant (Ld2 Lso1 Dg1 Dh++). The upper 20 cms were a very dark greyish brown (10YR 3/2) lake mud with abundant diatoms and a few plant remains.

The %dw profile (Figure 32) shows that %dw decreased progressively up the core from c. 28% at the core base to c.10% at the surface, probably related to changes in clay content. However, %loi remained low throughout the core at c.16-19% and exhibited little change except for a small increase in the upper 15 cm.

9.4 Radiometric Dating

The radiometric results suggest that sediments at this site have been subject to extensive focussing and vertical mixing. The ^{210}Pb and ^{137}Cs inventories are more than 5 times the values expected from atmospheric fallout, their activities are more or less uniform in the top 40 cm, and the depth of the 1963 level suggested by the ^{137}Cs record is significantly above the depth calculated by ^{210}Pb. In the event of mixing, the best guide to mean accumulation rates is the gradient of the ^{210}Pb profile beneath the mixed layer. Calculations based on this method give a value of 0.27±0.11 g cm^{-2}y^{-1}, compared to the ^{137}Cs derived value of 0.16±0.07 g cm^{-2}y^{-1}. In view of the uncertainties, the best approach would appear to use a mean accumulation rate of 0.22±0.09 g cm^{-2}y^{-1}, and this has been used to calculate the dates given in Table 9. The results are shown in Figure 33, Figure 34 and Figure 35.

Table 9 Chronology for Hatchet Pond

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0	0.00	1995	0				
10	1.29	1989	6	2			
20	3.15	1980	15	6			
30	5.02	1972	23	9			
40	7.12	1962	33	13	0.22	0.93	~40%
50	9.68	1950	45	18			
60	12.26	1938	57	23			
70	15.35	1924	71	29			
80	18.46	1909	86	35			

9.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in five levels of the sediment core were calculated and Figure 36 illustrates the results for the major taxa. Diatom preservation was good throughout the core. The diatom assemblages were very diverse. A total of 100 taxa was observed, 76 of which were present in the TP calibration set and 52 of which were present in the pH calibration set. Analogues for the TP reconstructions were generally good with more than 93% of the fossil assemblage used in the calibration procedure, except for the 70 cm sample where only 87% was used. Analogues were also good for the pH reconstructions with over 87% of the fossil assemblage being used in calibration, except for the 70 cm sample where only 82% could be used.

Figure 36 illustrates that there have been only small changes in the diatom species composition over the period represented by the core. The radiometric dating results indicated that the sediments have been subjected to focusing and vertical mixing and in view of the uncertainties with the profiles, only a mean accumulation rate of 0.93 cm yr^{-1} was used. This dates the 80 cm level to 1909 and therefore the whole core represents the post-1900 period.

All samples contained the same major taxa, although the assemblages were very diverse and none were dominated by a single species. The most common taxa were *Achnanthes minutissima, Synedra parasitica, Tabellaria flocculosa, Cyclotella stelligera, Fragilaria virescens* var. *exigua, Aulacoseira ambigua, Fragilaria pinnata* and *Fragilaria construens* var. *venter*. The only notable features were that *Aulacoseira ambigua, Achnanthes levanderi, Navicula jaernefeltii, Synedra acus* and *Achnanthes subatomoides* were present in higher relative abundances in the lower samples: 88 cm (c.1900), 70 cm (c.1924) and 50 cm (c.1950); and in contrast *Achnanthes minutissima, Synedra parasitica* and *Fragilaria construens* var. *venter* had higher frequencies in the upper samples: 30 cm (c.1972) and the surface. The diatom assemblages of Hatchet Pond were comprised, therefore, of both planktonic forms, e.g. *Aulacoseira ambigua* and *Cyclotella stelligera*, taxa commonly observed in mesotrophic waters, and non-planktonic forms, e.g. the benthic *Fragilaria* spp. and *Achnanthes minutissima*, taxa frequently observed across a range of lake types but commonly found in shallow, non-acid waters.

9.6 Total Phosphorus and pH Reconstructions

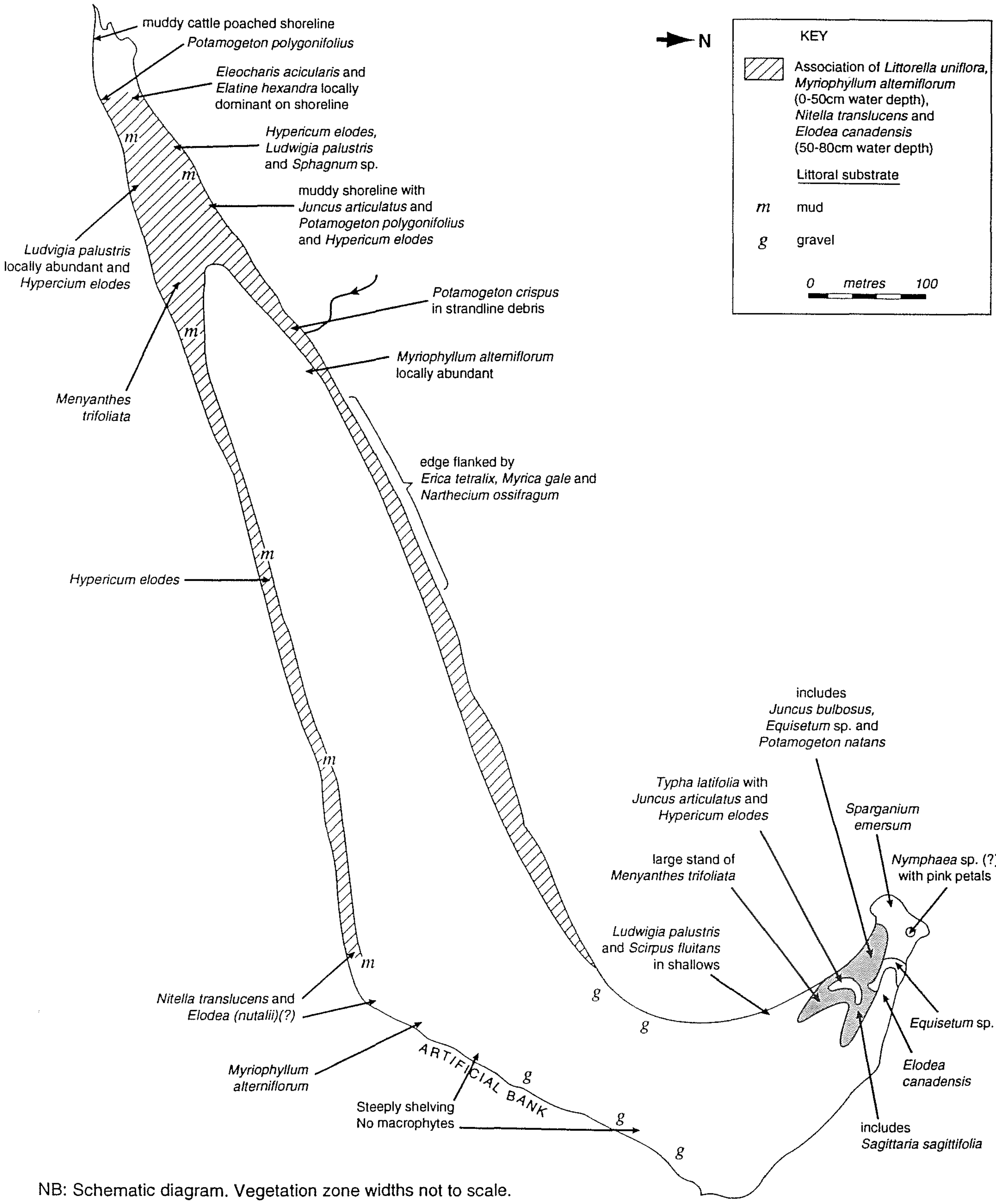
The TP reconstruction shows that there has been a slight increase in the nutrient status of Hatchet Pond since c. 1972. The values produced by the model were 23 µg l^{-1} for c.1900 (88 cm), 18 µg TP l^{-1} for c.1924 (70 cm) although this lower value is probably due to the poorer species analogues in this sample, 25 µg TP l^{-1} for c.1950 (50 cm) and c.1972 (30 cm), increasing slightly to 33 µg TP l^{-1} in the 1995 sample. The higher TP value for the surface sample is explained by the higher relative abundance of *Fragilaria construens* var. *venter*, which has an intermediate TP optimum and a wide TP tolerance.

The pH reconstruction shows that there has been a slight increase in pH. The DI-pH values were 6.2 for c.1900, 6.1 for c.1924 (although there were poor analogues for this sample), increasing to 6.5 for c.1950, c.1972 and 1995. This inferred pH increase appears to be related to the increasing importance of diatom species less tolerant of acid waters in the upper samples.

9.7 Discussion

Hatchet Pond appears to have been a mesotrophic lake since at least 1900 with evidence of slight enrichment in the last two decades. Unfortunately, there are few available water chemistry data with which to compare the diatom model results, although the site is described as a mesotrophic one in the conservation literature. In a survey carried out by NRA Southern Region of Hatchet Stream from 1979 to 1983, orthophosphate ranged from <5 µg P l^{-1} to 30 µg P l^{-1} with a data set mean of 11 µg P l^{-1} but TP was not measured. However, given that TP would be slightly higher than these values, the DI-TP results appear to be in the correct range. The pH model indicates that the lake has become more base-rich since c.1950 but it seems to slightly under-estimate current pH values. The NRA survey recorded pH in the range 6.4 to 8.5 with a data set mean of 7.2 whereas the DI-pH values were c. 6.5. This under-estimation by the model is probably due to the large number of acid lakes in the SWAP training set and very few sites with diatom assemblages analogous to those of Hatchet Pond (Stevenson *et al.*, 1991).

Figure 31 Aquatic macrophyte distribution map for Hatchet Pond



NB: Schematic diagram. Vegetation zone widths not to scale.

Figure 32 Lithostratigraphic data for Hatchet Pond

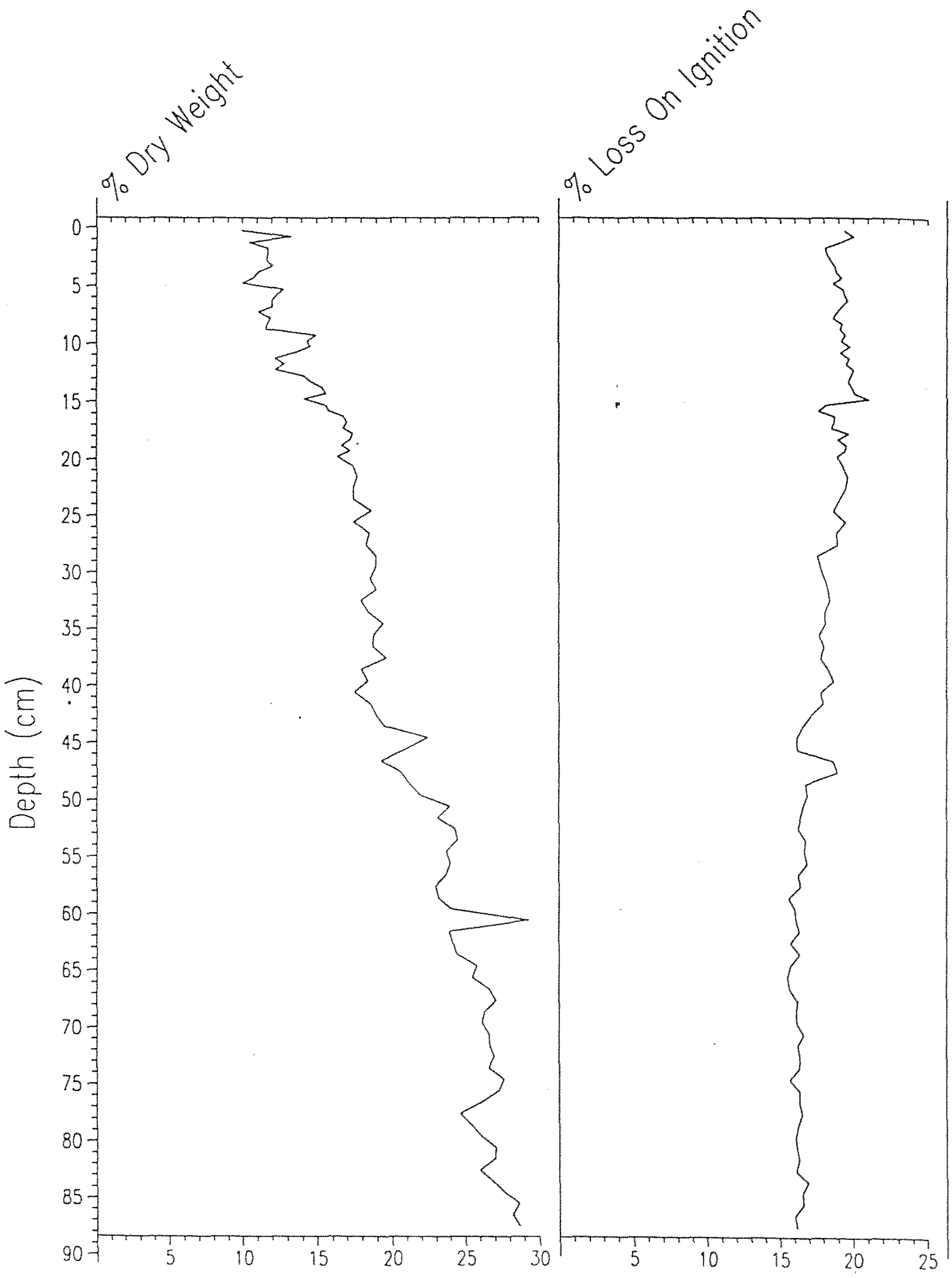


Figure 33 ^{210}Pb Activity versus Depth - Hatchet Pond

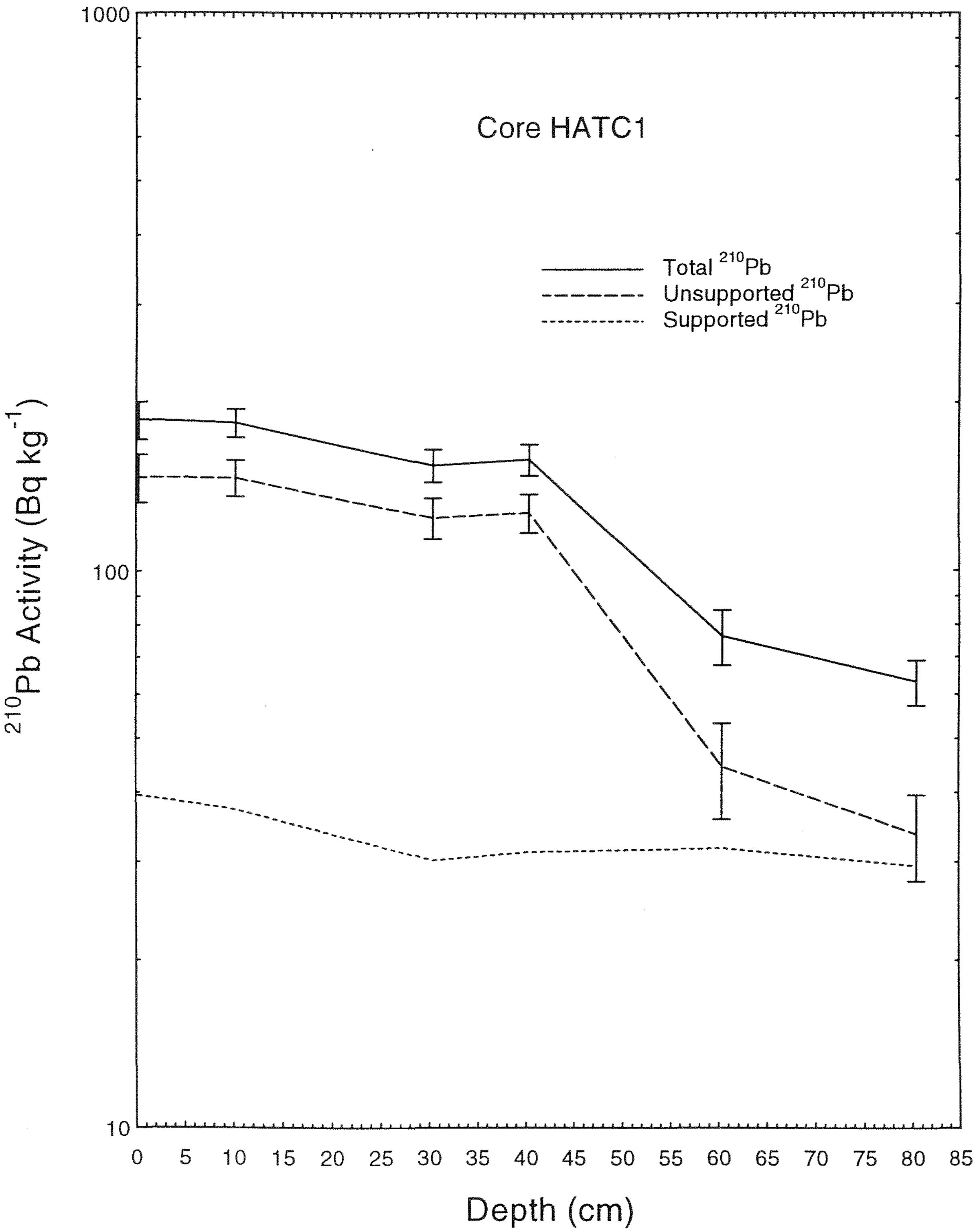
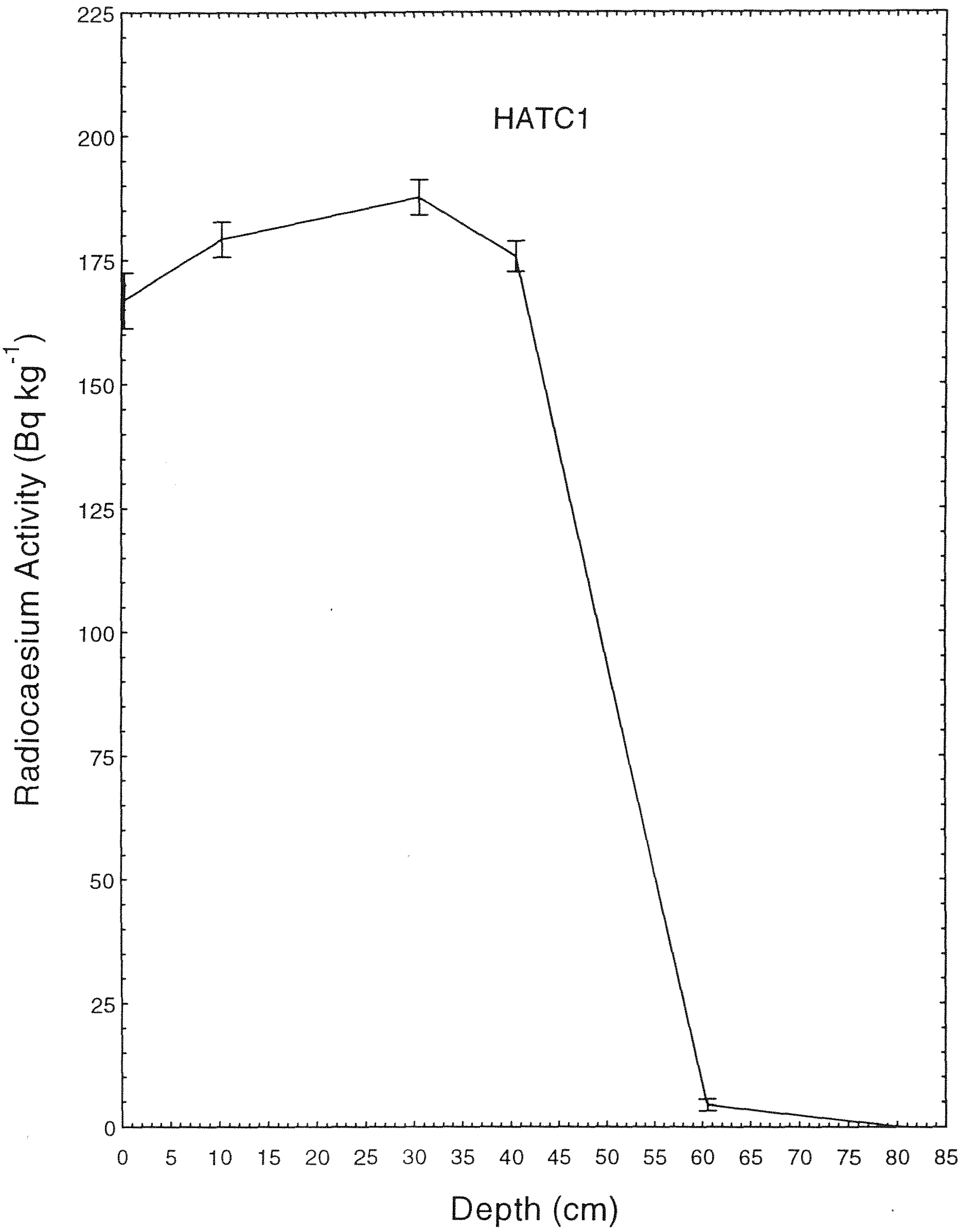


Figure 34 ^{137}Cs Activity versus Depth - Hatchet Pond



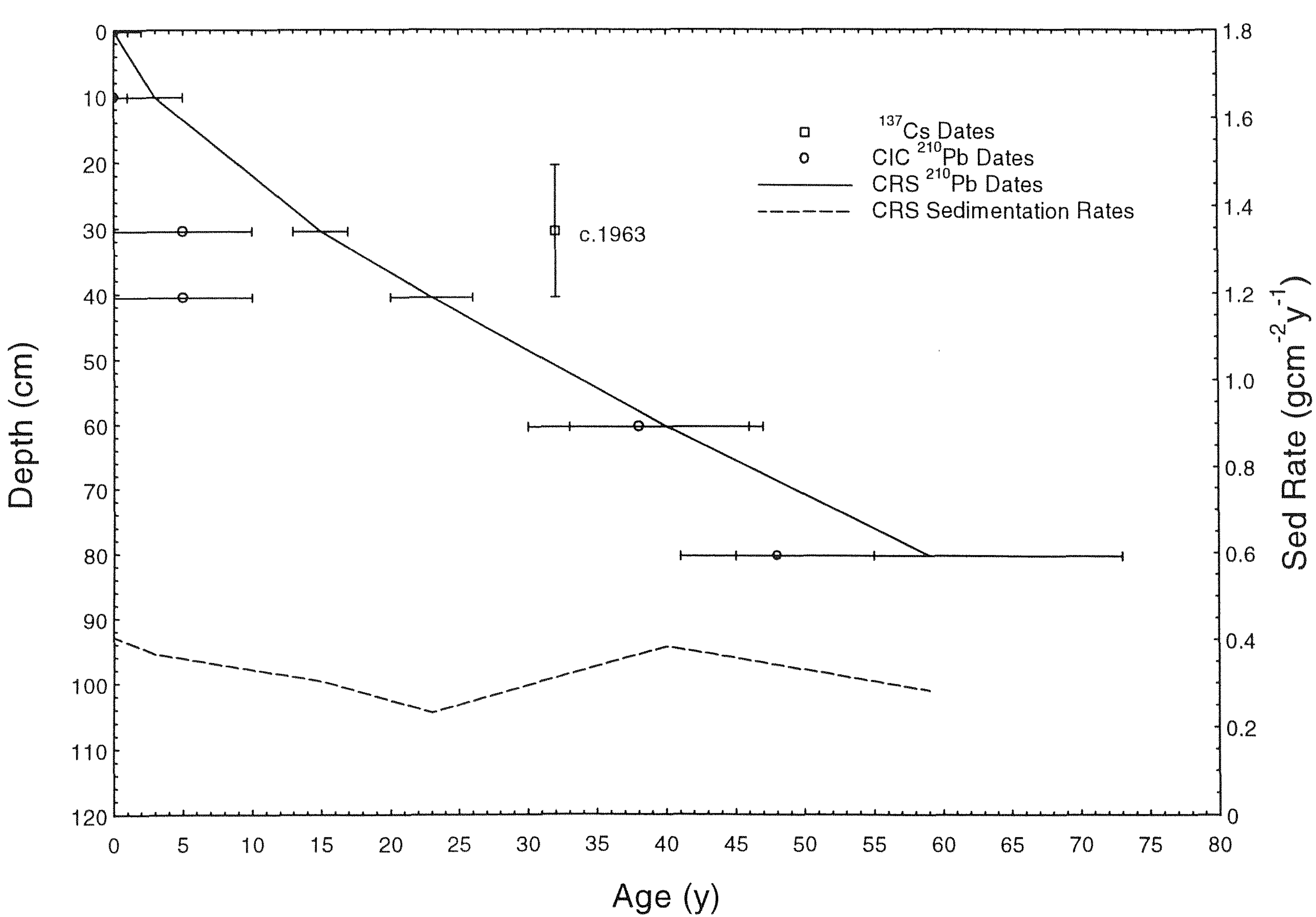
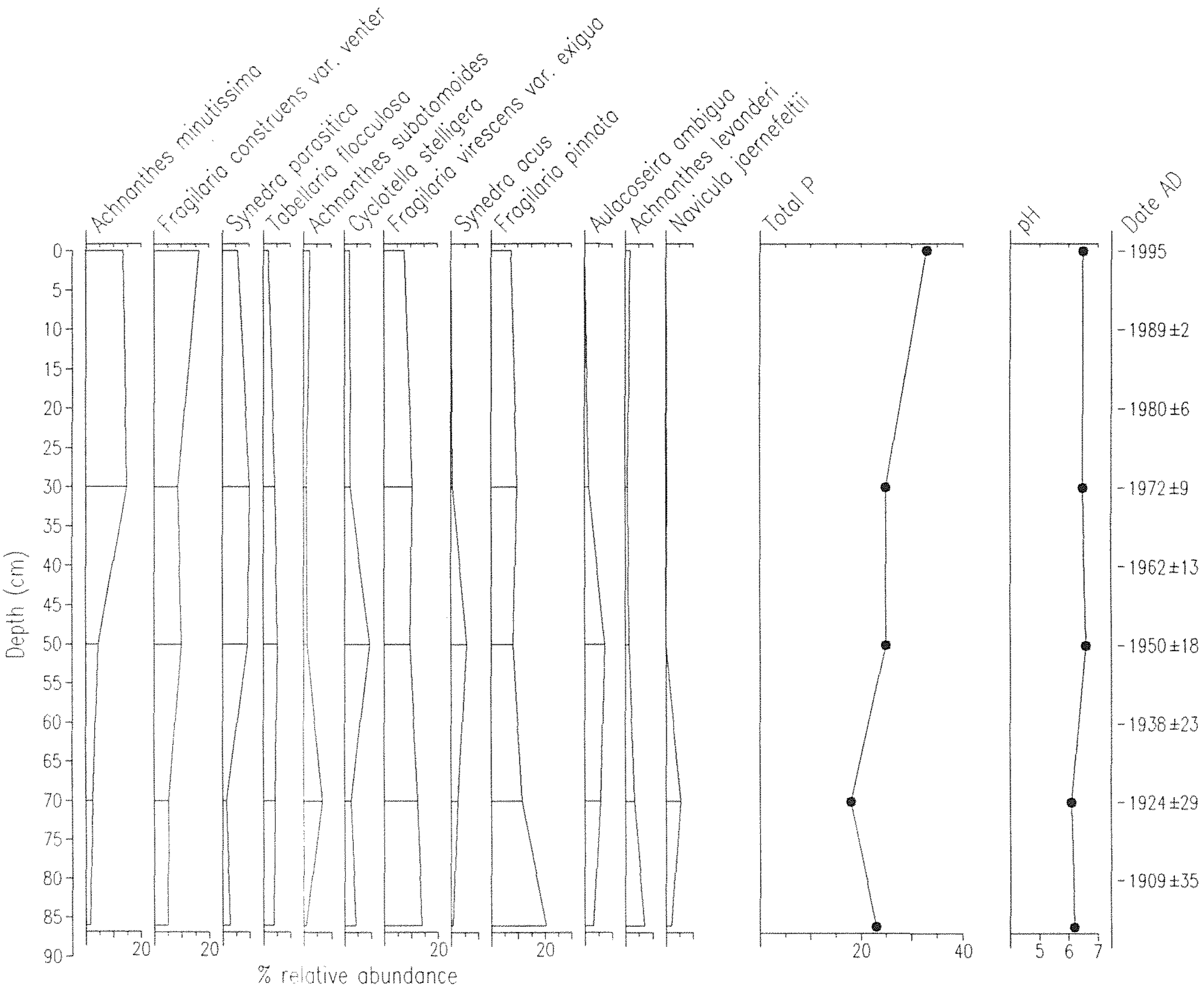


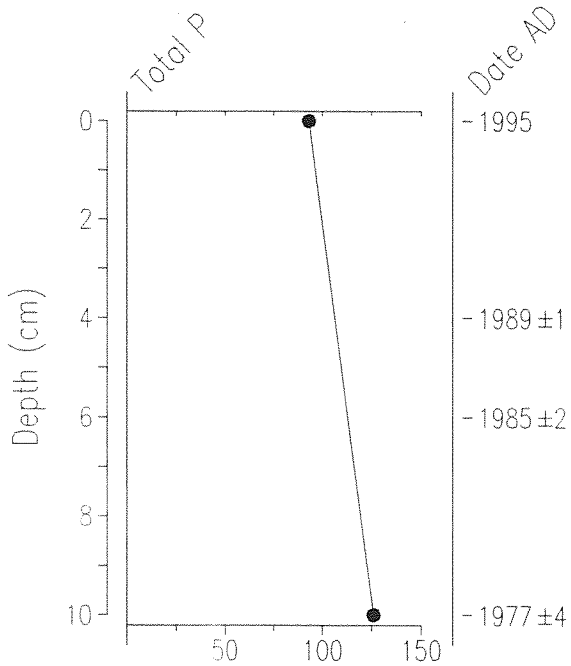
Figure 35 Depth versus Age - Hatchet Pond

Figure 36 Summary diatom diagram and reconstructions for Hatchet Pond



Semerwater

- **A palaeolimnological interpretation of events at Semerwater is severely limited by diatom preservation problems.**



µg TP l^{-1}

10. Semerwater, North Yorkshire (SD 918 874)

10.1 Site Description and Water Chemistry

- Semerwater is an upland, natural, lime-rich lake situated in the Yorkshire Dales
- Altitude 250 m
- Mean depth c. 4 m
- Maximum depth 12 m
- It is fed by precipitious streams and is subject to large fluctuations in water level
- Land use is c. 70% pasture (some improved); 20% hay meadows and 10% forestry and mixed woodland.

pH	units	7.8
conductivity	$\mu S\ cm^{-1}$	218
alkalinity	mg $CaCO_3$	90
calcium	mg l^{-1}	38.5
total nitrogen	mg l^{-1}	2.0
total phosphorus	$\mu g\ l^{-1}$	30

nb. These readings were taken from the lake outflow

10.2 Macrophyte Survey

Date of visit 9-6-95.
An aquatic macrophyte distribution map is presented in Figure 37 and the transect profile is shown in Figure 38. Water clarity was poor on the day of survey and a secchi depth of 1.3m was recorded. The water level appeared to be approximately 0.3m below normal.

The east shoreline is characterised by a cobble/pebble substrate, and very few macrophytes are evident within the littoral zone except for *Fontinalis antipyretica* attached to the cobbles. *Caltha palustris* and *Myosotis scorpiodes* were occasional above the normal waterline and *Juncus effusus* was abundant at the pasture/shoreline fringe. A substantial stand of *Nuphar lutea* occupies the south-east corner of the lake, behind which the shoreline soils were severely poached by cattle.

The southern shoreline substrate consists chiefly of sand and a significant area is covered in a lawn of *Eleocharis acicularis* to a water depth of approximately 0.2m. *Potamogeton pectinatus* is also found here in local abundance to a depth of approximately 0.3m.

The west shore supports areas of sedge fen, with several stands of *Carex rostrata*, often bordered on the open water side by *Scirpus lacustris* ssp. *lacustris* and *Equisetum fluviatile. Potamogeton pectinatus* is the most commonly occurring submerged macrophyte in shallow water on this side of the lake. *Elodea canadensis*, *Chara* sp., *Calliergon* sp. and *Callitriche stagnalis* are also found here, although the deeper water benthic habitat (from approximately 0.7 - 1.4m) is largely dominated by the filamentous alga *Cladophora* sp..

Open water in the bay leading to the outflow is dominated by *P. pectinatus*, and *Chara* sp. is also present at approximately 0.2m water depth In open water, *Cladophora* sp. generally blankets sediments in a zone from 0.5 - 1.4 m depth which represents a large proportion of the total lake surface area.

No past aquatic macrophyte data was available for Semerwater although Ratcliffe (1977) mentions that emergent and submerged vegetation is sparse. The apparently impoverished plant assemblage of Semerwater is typical of lakes which have undergone significant eutrophication. *Potamogeton pectinatus*, and the alga *Cladophora* sp. for example, tend to thrive at sites which have become nutrient enriched and it seems unlikely that these species would have been the dominant taxa before the time of agricultural intensification. Applying the species list presented in Table 10, Semerwater is classified as Type 10B (Palmer, 1992), a eutrophic category.

Table 10 Species list and DAFOR abundance rating for Semerwater

	abundance	notes
Submerged taxa		
Cladophora sp.	A	dominant within middle-depth zone
Chara sp.	O	in shallow water in west
Calliergon sp.	R	in shallow water in east
Potamogeton pectinatus	F	locally abundant in north
Elodea canadensis	R	in shallow water in west
Fontinalis antipyretica	F	on east shoreline
Eleocharis acicularis	O	dominant on sand by main inflow
Callitriche stagnalis	R	near minor inflow
Floating leaved taxa		
Nuphar lutea	R	locally abundant in south-east
Emergent taxa		
Phalaris arundincea	O	in west
Scirpus lacustris ssp. *lacustris*	F	in west
Equisetum fluviatile	O	locally abundant in north-west
Carex rostrata	F	locally abundant in west
Carex vesicaria	O	in west
Caltha palustris	F	on east shoreline
Salix sp.	A	dominant behind sedge fringe in west
Eleocharis palustris	R	in north west
Myosotis scorpiodes	O	on north-east shore
Juncus effusus	F	

10.3 Lithostratigraphy

A 75 cm sediment core (SEME1) was taken from the deepest part of the lake at a water depth of 11.5 m using a Mackereth corer on 9-6-95. The clay-silt content of the sediment decreased up the core from a very dark grey (10YR 3/1) inorganic clay with silt and no detritus between the core base and 30 cm (As2 Ag1 Ld1), to a lake mud with some silt and clay and plant remains between 25 and 5 cm (Ld2 Ag2 As+ Dh+ Lso+). The upper 5 cms were a very dark brown (10YR 2/2), organic, silty lake mud with fragments of herbaceous plants (Ld3 Ag1 Lso+ Dh++).

The %dw and %loi profiles (Figure 39) show a clear and gradual increase in percentage organic matter from the core base to the surface, %loi rising from 10% to 18% and %dw decreasing from 52% to 20%. Wd follows the same trend decreasing from 1.45 g cm^{-3} at the core base to 1.10 g cm^{-3} at the surface.

10.4 Radiometric Dating

Although the Semerwater core appeared to have a relatively simple ^{210}Pb profile, there were significant differences between dates calculated using the CRS and CIC ^{210}Pb dating models due to an abrupt termination of the unsupported ^{210}Pb record at c.25 cm. Whereas the CIC model indicates a more or less constant sedimentation rate for the past 40 years of 0.15±0.03 g $cm^{-2}y^{-1}$, the CRS model suggests much lower accumulation rates prior to the past 20 years. The hiatus at 25 cm appears to be associated with a shift from less dense sediments above this level to more dense sediments in the deeper sections.

The ^{137}Cs profile has a well defined peak at 18±2.5 cm that appears to record the 1963 weapons test fallout maximum. Within the limits of the resolution of the ^{137}Cs peak this date is in reasonable agreement with the CIC model ^{210}Pb results, supporting the inference of an incomplete ^{210}Pb record.

The results obtained are summarised in Table 11 and Figure 40, Figure 41 and Figure 42. Dates above the apparent ^{210}Pb equilibrium level have been calculated using the mean sediment accumulation rate of 0.17±0.04 g $cm^{-2}y^{-1}$ determined by the CIC model and the ^{137}Cs record. Also shown are depths of the 1930, 1900 and 1850 levels calculated by extrapolation of the ^{210}Pb results though these should be regarded with some caution in view of the possible hiatus at 25 cm.

Table 11 Chronology of Semerwater

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0	0.00	1995	0				
2	0.52	1992	3	1		0.64	
4	1.07	1989	6	1		0.60	
6	1.66	1985	10	2		0.54	
8	2.33	1981	14	3		0.51	
10	3.00	1977	18	4	0.17	0.50	23.5
12	3.69	1973	22	5		0.49	
14	4.39	1969	26	6		0.48	
16	5.12	1965	30	7		0.46	
18	5.87	1960	35	8		0.45	
20	6.63	1956	39	9		0.44	

Extrapolated levels:

1930	30 cm
1900	38 cm
1850	51 cm

10.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in only two levels of the sediment core were calculated and Figure 43 illustrates the results for the major taxa. Diatom preservation was extremely poor throughout the core with dissolution and breakage problems, and counting was difficult due to low concentrations and interference from mineral matter. Diatom (silica) dissolution is not uncommon in lime-rich waters. Therefore, it was only possible to analyse the upper 10 cms. A total of 75 taxa was observed, 62 of which were present in the calibration set. Most of the common taxa were well represented in the calibration set, with greater than 93% of the fossil assemblage being used in the calibration procedure.

Figure 43 illustrates that there is little difference between the diatom species composition of the 10 cm sample (1977) and that of the surface sample. The results must, however, be regarded with some caution in view of the dissolution problems. For example the lower percentages of *Stephanodiscus hantzschii* relative to *Fragilaria pinnata* in the 10 cm sample compared with those in the surface sample could be explained by the fact that *S. hantzschii* is more lightly silicified than *F. pinnata* and is thus more susceptible to dissolution. The assemblages of both levels were dominated by non-planktonic taxa associated with plant or stony substrates, *S. hantzschii* being the only important planktonic species. These taxa are commonly observed in shallow, eutrophic waters.

10.6 Total Phosphorus Reconstruction

The TP reconstruction should be interpreted with caution due to the problems discussed above. The diatom model provides an estimate of 126 μg TP l^{-1} for 1977 (10 cm) and 93 μg TP l^{-1} for 1995.

10.7 Discussion

A palaeolimnological interpretation of events at Semerwater is clearly limited by the diatom preservation problems. However, the results indicate that Semerwater has not experienced any major changes in TP concentrations in the last two decades. The diatom-inferred values closely match the measured TP concentrations of the lake from a survey in 1990 (Gibbons, 1990) where mean TP was 80 μg TP l^{-1} and maximum TP was 125 μg TP l^{-1}. These values place the lake in the strongly eutrophic category. The same study, however, showed that TP concentrations in the main inflow (Crooks Beck) were only 10 μg TP l^{-1} and the recent EA readings from the lake outflow produced a mean value of 30 μg TP l^{-1} over the period July 1995 to February 1996. These data suggest that the high TP in the lake may be internally derived, most likely from sediment resuspension, but a detailed nutrient budget would need to be calculated to draw any firm conclusions. Magnetic measurement research on a number of deep water cores is ongoing at the University of Edinburgh, which should allow the calculation of whole lake sediment fluxes and hence catchment erosion rates (R. Thompson, pers. comm.).

Screening of a sediment core from Semerwater taken by the University of Edinburgh revealed similar problems of dissolution and low diatom concentrations (V. Jones, unpublished). However, diatoms were preserved enough to compile species lists and this work showed that samples below 70 cm were dominated by benthic taxa, particularly *Fragilaria brevistriata* whilst the upper samples were dominated by the same taxa as those found in SEME1, eg. *Stephanodiscus* spp., *Achnanthes* spp., and *Amphora pediculus*. A switch from a benthic flora to one where planktonic taxa such as *Stephanodiscus* spp. are also important could signal reduced light conditions brought about by increased nutrient concentrations but these findings cannot be confirmed by the current study owing to poor diatom preservation.

Figure 37 Aquatic macrophyte distribution map for Semerwater

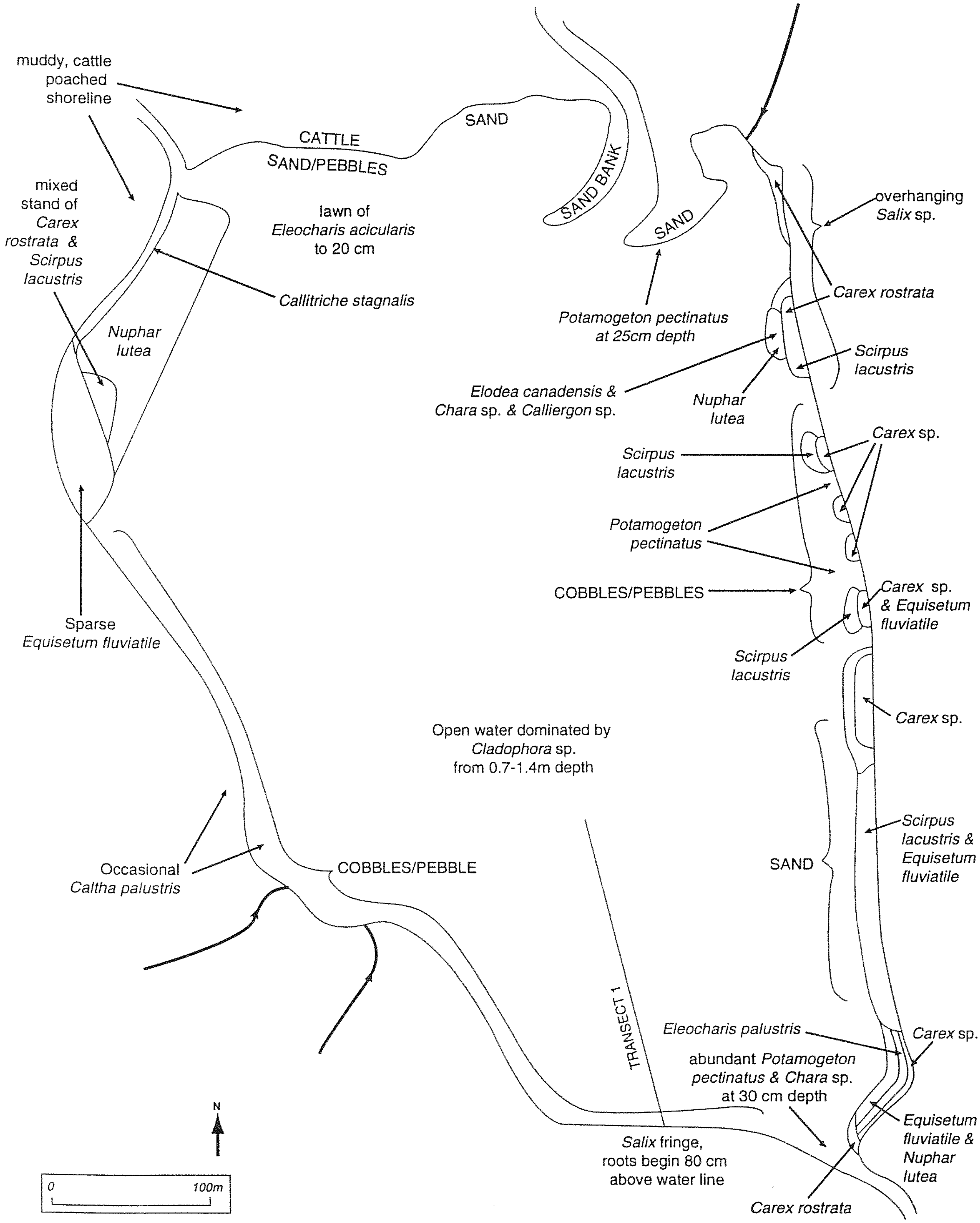


Figure 38 Aquatic macrophyte transect profile for Semerwater

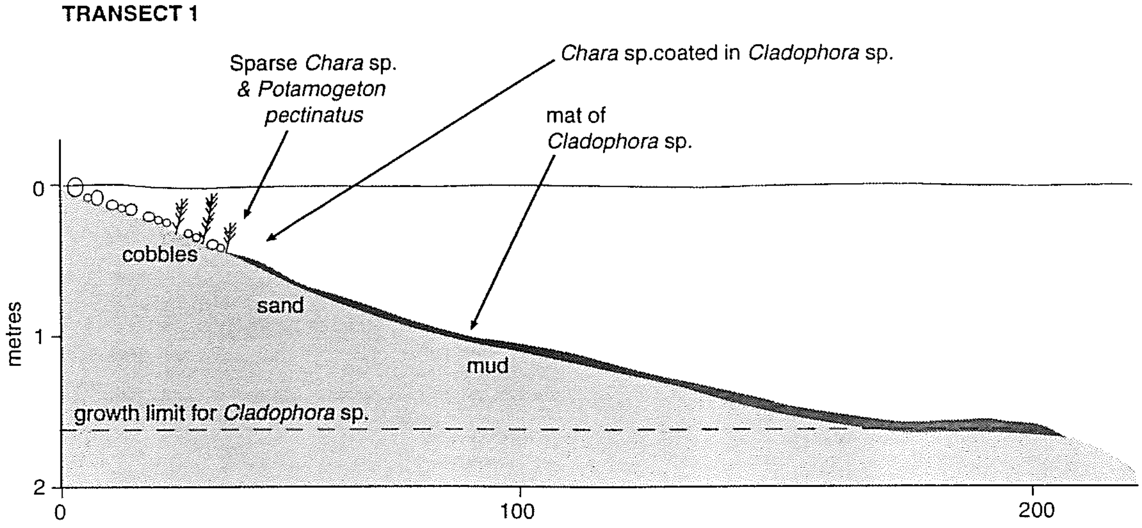


Figure 39 Lithostratigraphic data for Semerwater

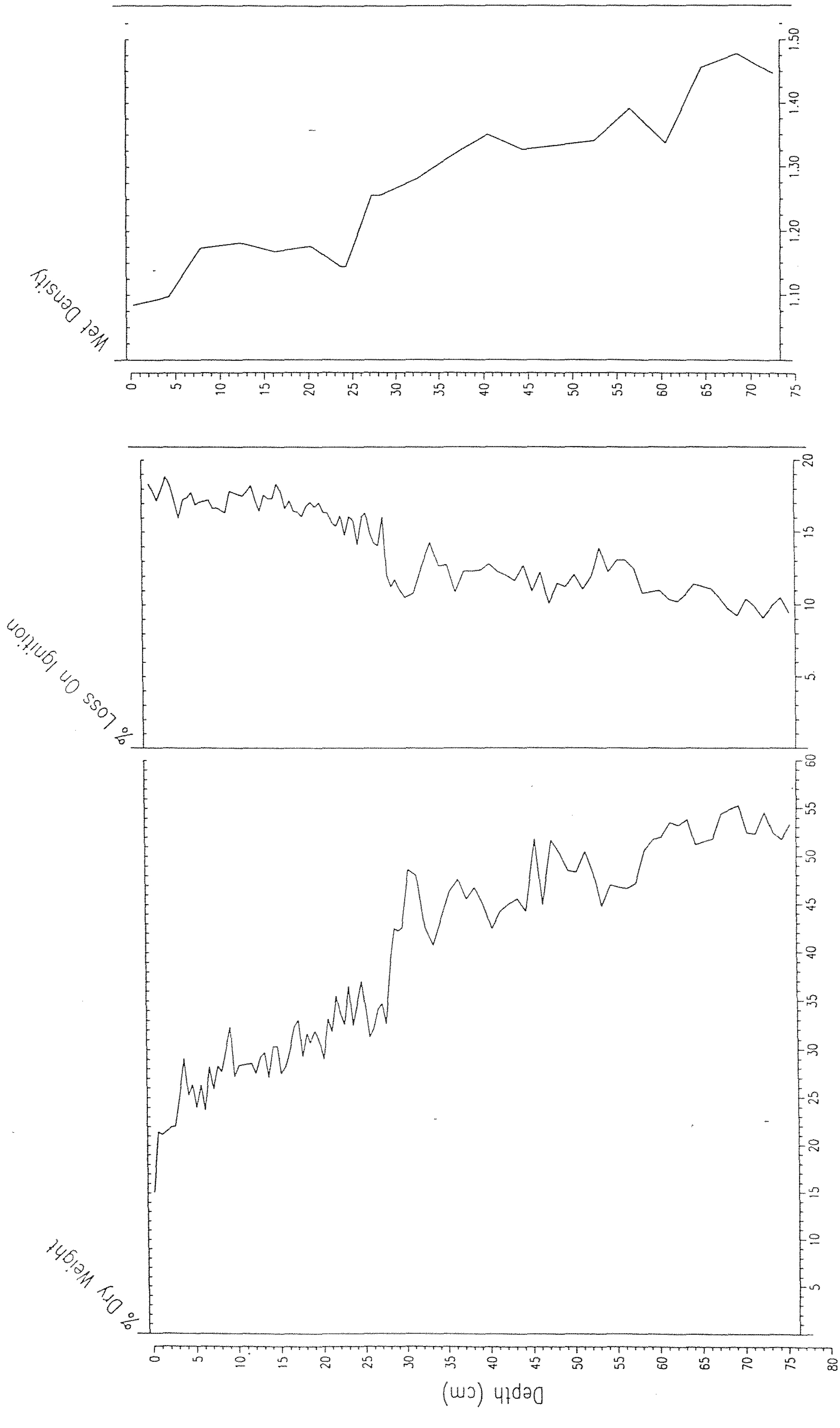


Figure 40 ^{210}Pb Activity versus Depth - Semerwater

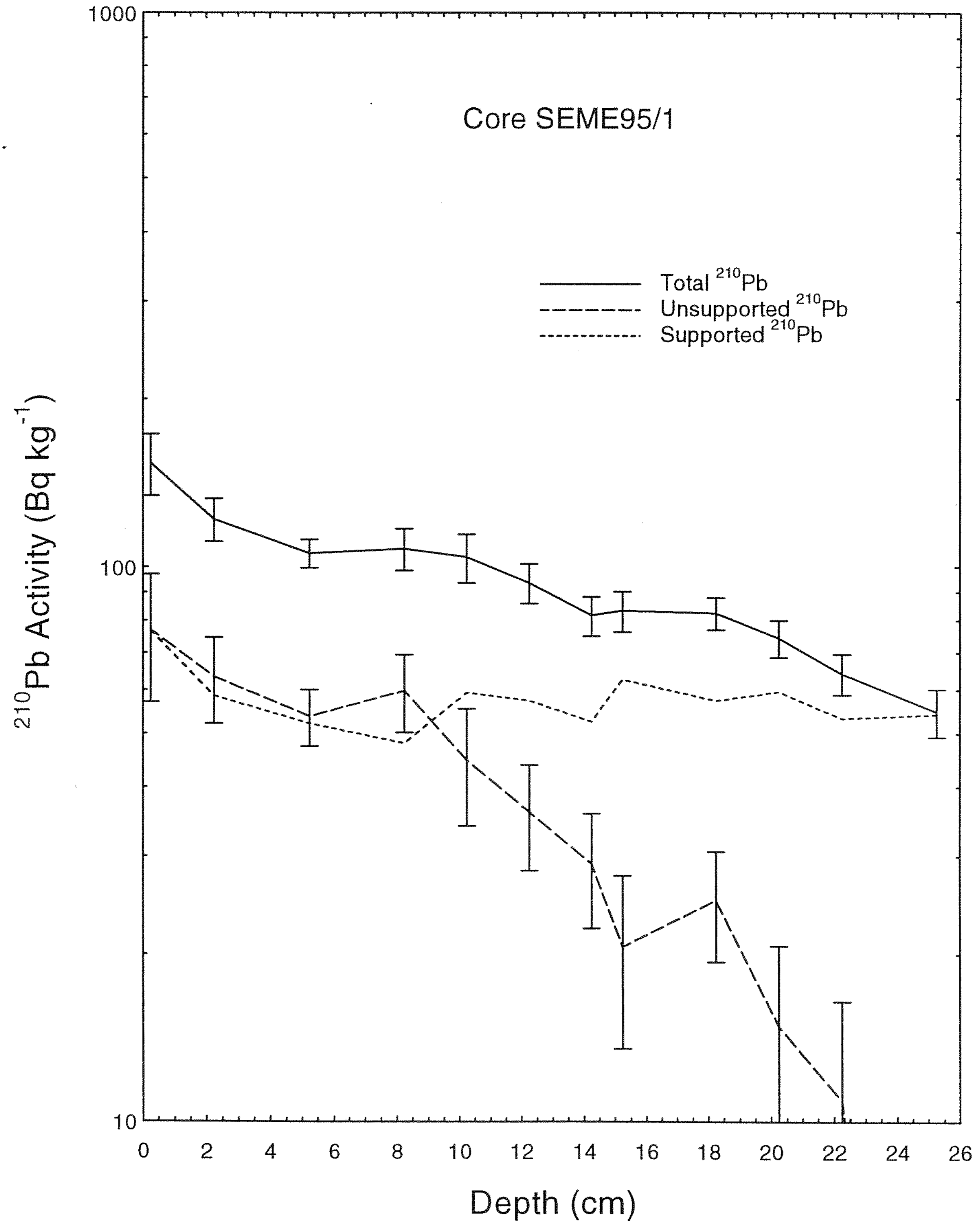


Figure 41 ^{137}Cs and ^{241}Am Activity versus Depth - Semerwater

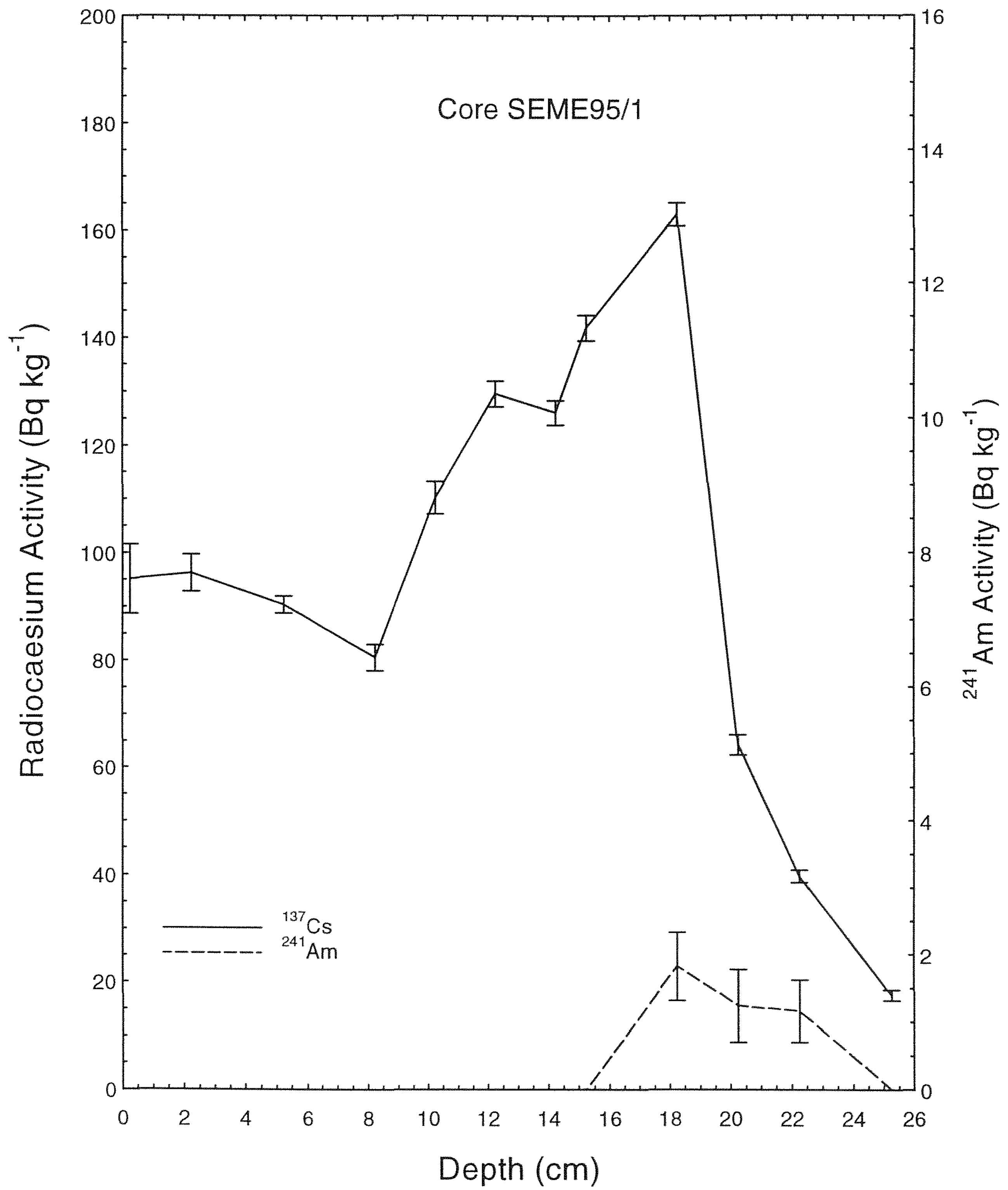


Figure 42 Depth versus Age - Semerwater

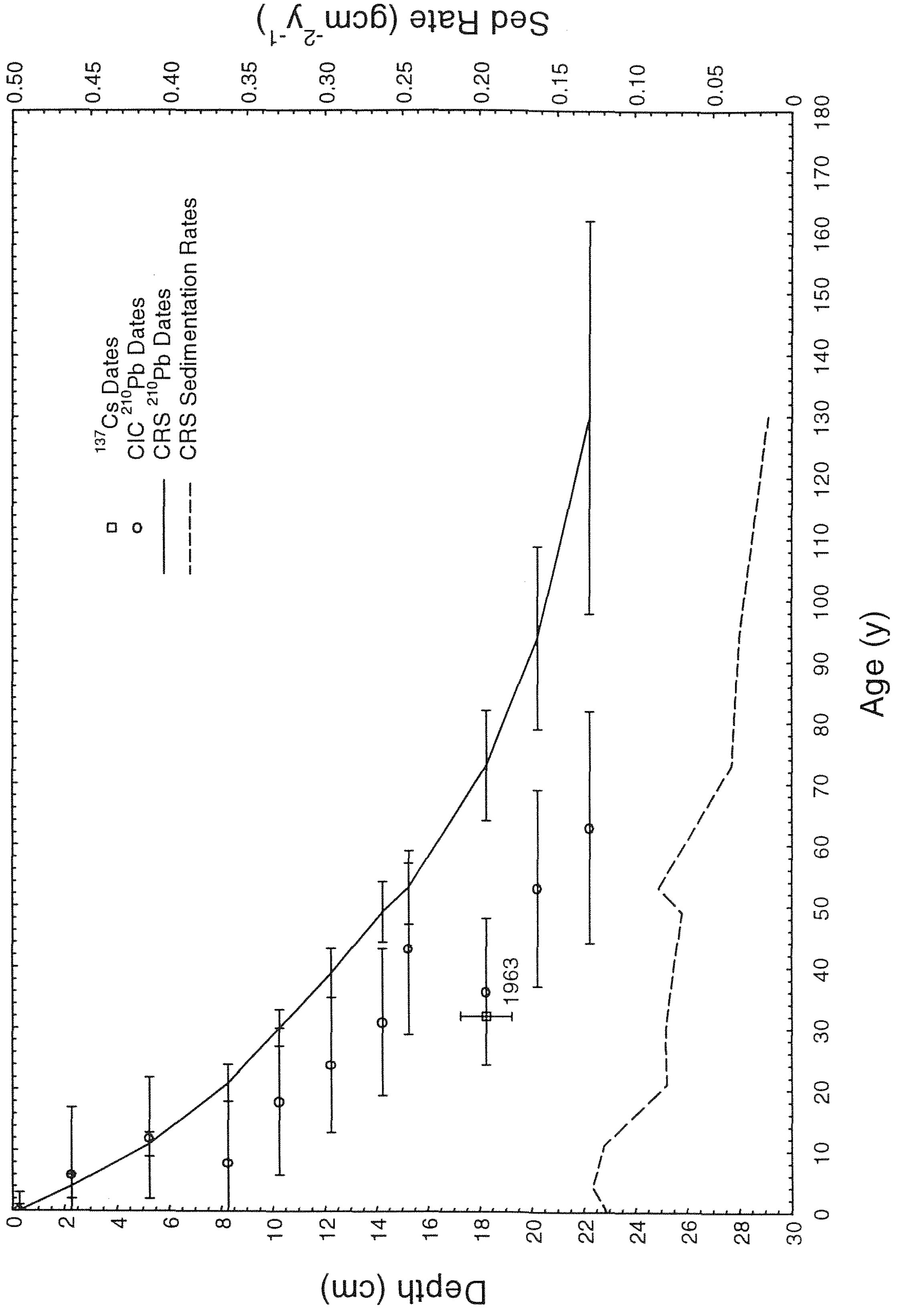
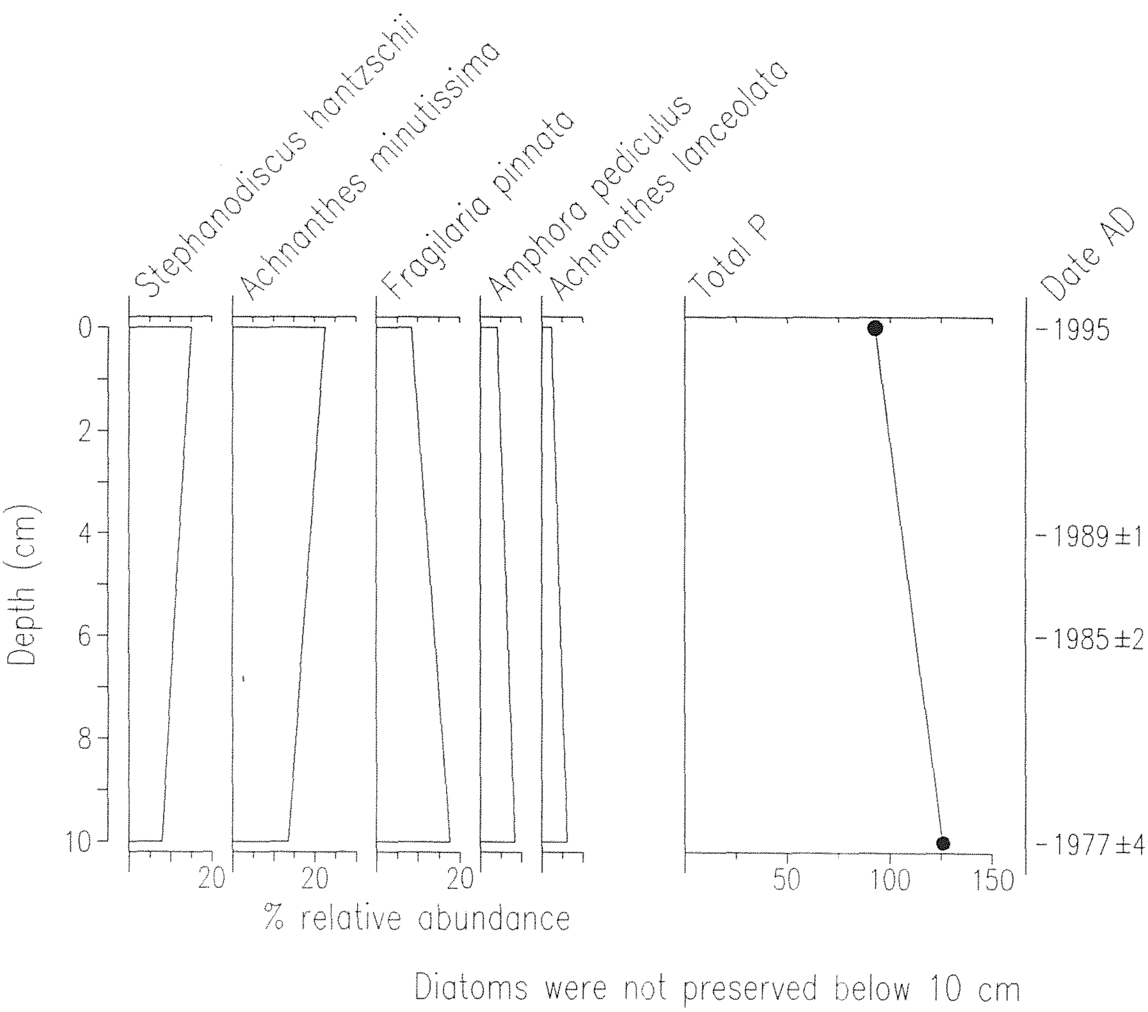
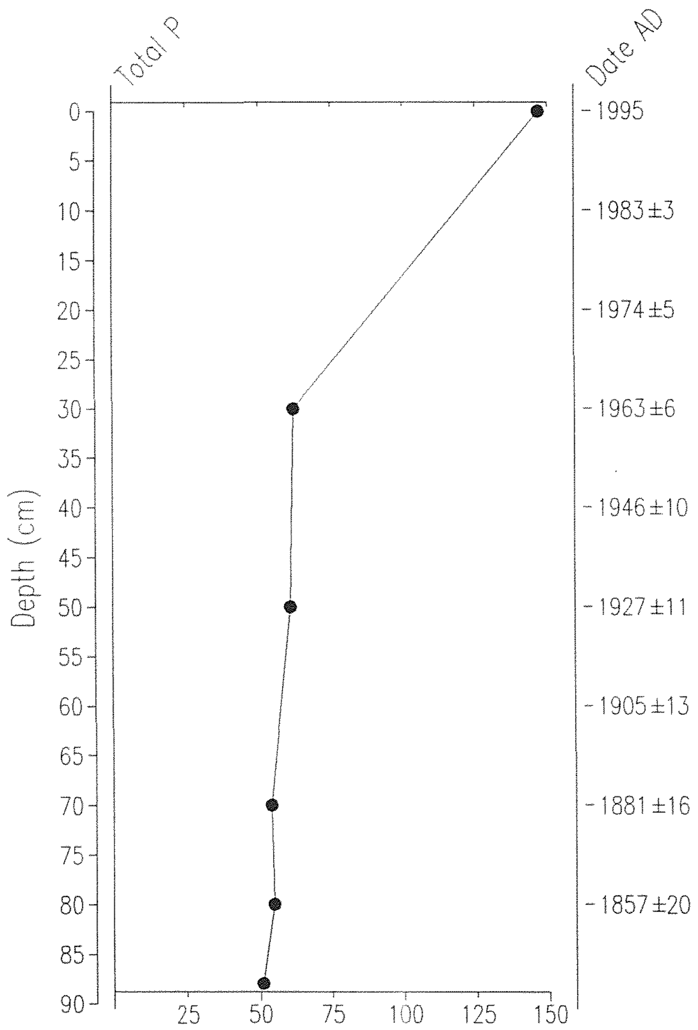


Figure 43 Summary diatom diagram and reconstruction for Semerwater



Betton Pool

- **The DI-TP results indicate that Betton Pool has been eutrophic since at least 1850, but that the lake has experienced significant enrichment since 1960 and would now be classified as hypertrophic.**
- **Betton Pool appears to be a naturally eutrophic lake with TP concentrations similar to many of the Cheshire and Shropshire meres, but seems to have been affected by anthropogenic impacts in recent decades, such as agricultural intensification and recreation.**



µg TP l^{-1}

11. Betton Pool, Shropshire (SJ 509 078)

11.1 Site Description and Water Chemistry

- Altitude 85 m
- Area 6.4 ha
- Maximum depth 11 m
- Fed largely by groundwater
- Sandy substrates
- The catchment is predominantly arable with some scattered trees and a strip of mixed woodland at the southern end. There is some ploughing near the water's edge.

pH	units	8.0
conductivity	$\mu S\ cm^{-1}$	306
alkalinity	mg $CaCO_3$	98
calcium	mg l^{-1}	34.5
total nitrogen	mg l^{-1}	1.8
total phosphorus	$\mu g\ l^{-1}$	87

11.2 Macrophyte Survey

Due to unforeseen circumstances, a macrophyte survey was not conducted at Betton Pool in summer 1996. The survey has been rescheduled for summer 1997 and therefore, the results can unfortunately not be reported here. The results will, however, be forwarded on completion.

11.3 Lithostratigraphy

A 90 cm sediment core (SCM27B) was taken from the deepest part of the lake in a water depth of 11 m using a Mackereth corer on 15-2-95. The sediment was a very dark greyish brown (2.5Y 3/2) from 25 cm to the core base. The lower sediments (> 60 cm) were a silty lake mud with a reasonable clay content and some plant remains. The sediment appeared to have less clay content above 60 cm. The upper 25 cms were dark yellowish brown (10YR 3/4) but sediment composition was similar to the lower part of the core.

The %dw profile (Figure 44) shows that there has been a small but steady decrease in %dw up the core from c. 20% at the base to c. 10% at the surface. The %loi values have increased from c. 20% at the core bottom, reached a maximum of c. 30% between 25-45 cm and then declined again to c. 25% in the upper 25 cms, indicating a period of more organic sediments in the central core section.

11.4 Radiometric Dating

The ^{210}Pb results indicate rapid sedimentation, equilibrium with the supporting ^{226}Ra being reached at a depth of more than 70 cm. The non-monotonic feature at 20-25 cm indicates a major irregularity in the process of sediment accumulation and there is in consequence a significant discrepancy between CRS and CIC model ^{210}Pb dates for this part of the core. The 1963 level in the core is clearly marked by a well resolved ^{137}Cs peak at 30±5 cm depth, in good agreement with the CRS model results, and these have accordingly been used to calculate the core chronology given in Table 12.

The ^{210}Pb results suggest that the irregularity at 20-25 cm records an episode of rapid sedimentation during the early 1970s, but that excluding this event there was a more or less uniform sedimentation rate of 0.10±02 g $cm^{-2}y^{-1}$. Extrapolated dates below the ^{210}Pb dating horizon have been calculated using this value. The results are illustrated in Figure 45, Figure 46 and Figure 47.

Table 12 Chronology of Betton Pool

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0	0.00	1995	0				
5	0.76	1989	6	2	0.135	0.82	27.3
10	1.53	1983	12	3	0.163	0.95	32.2
15	2.47	1978	17	4	0.209	1.18	36.5
20	3.42	1974	21	5	0.255	1.41	40.8
25	4.29	1969	26	6	0.192	1.07	36.8
30	5.16	1963	32	6	0.116	0.67	31.9
35	6.03	1955	40	8	0.097	0.53	32.9
40	6.91	1946	49	10	0.084	0.43	34.6
50	8.80	1927	68	11	0.10	0.45	
60	11.00	1905	90	13	0.10	0.45	
70	13.37	1881	114	16	0.10	0.45	
80	15.75	1857	138	20	0.10	0.45	

11.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in six levels of the sediment core were calculated and Figure 48 illustrates the results for the major taxa. Diatom preservation was rather poor throughout the core. A total of 89 taxa was observed, 73 of which were present in the calibration set. All of the common taxa were well represented in the calibration set, with greater than 96% of the fossil assemblage being used in the calibration procedure.

Figure 48 illustrates that there have been marked changes in the diatom species composition over the period represented by the core (post-1850). The three lower samples: 88 cm (c.1850), 80 cm (c.1857), and 70 cm (c.1880) were all dominated by mesotrophic, planktonic taxa, particularly *Cyclotella ocellata*, *Cyclotella radiosa, Stephanodiscus alpinus* and two non-planktonic forms, *Achnanthes lanceolata* and *Fragilaria pinnata.* There was a change by the 50 cm sample (c. 1927) where *Cyclotella ocellata* disappeared and *Aulacoseira subarctica, Stephanodiscus parvus* and *Cyclostephanos dubius* appeared for the first time, the latter two taxa being associated with enriched waters. The assemblage in the 30 cm sample (c.1963) was similar to the 50 cm level except that *Fragilaria crotonensis* , a planktonic species commonly found in nutrient-rich waters, was observed for the first time. The surface sample was markedly different from the 30 cm sample. The mesotrophic taxa decreased in relative abundance, particularly *Cyclotella radiosa, Aulacoseira subarctica* and *Stephanodiscus alpinus.* Conversely, the relative abundances of *Fragilaria crotonensis* and in particular *Stephanodiscus parvus* (55%) increased.

11.6 Total Phosphorus Reconstruction

The TP reconstruction indicates that the lake has been eutrophic in terms of TP concentrations since at least 1850. The model infers that TP concentrations increased slightly between 1880 and c.1930 but that the lake has experienced significant enrichment since 1960. For example, the DI-TP concentrations were 51 μg TP l^{-1} in c.1850 (88 cm), 55 μg TP l^{-1} in 1857 (80 cm), 54 μg TP l^{-1} in 1880 (70 cm), increasing slightly to 61 μg TP l^{-1} in 1927 (50 cm), 62 μg TP l^{-1} in 1963 (30 cm) and then more than doubled to 147 μg TP l^{-1} by 1995, related to the expansion of *Stephanodiscus parvus*, a small, centric planktonic diatom commonly observed in highly enriched lakes. This long term stability followed by a recent increase in productivity is supported by the ^{210}Pb profile, which suggested an episode of rapid increase in sediment accumulation rates in the early 1970s following a long period of more or less uniform accumulation.

11.7 Discussion

The significant recent eutrophication at Betton Pool, as inferred from the diatom model, is consistent with available documented data on land use and water quality. The lake is surrounded by arable fields with ploughing near the waters edge and fields used for cattle grazing, and is also used for water ski-ing and angling. The lake also appears to have been used as a roost by a large gull population since the late 1980s, attracted by an adjacent refuse site. All of these catchment and lake uses act as potential sources of nutrients and have all either increased (e.g. agricultural intensification) or been introduced (e.g. recreation) in recent decades.

The inferred TP concentration of 147 μg TP l^{-1} for 1995 closely matches recent water chemistry data collected during 1991/2, where annual mean TP was 113 μg TP l^{-1} and ranged from 8-200 μg TP l^{-1} over the sampling period (Moss *et al.*, 1992). Likewise in the 1995/6 EA survey, TP in the lake ranged from 23 to 211 μg TP l^{-1} . Although high, placing Betton Pool in the eutrophic category, these concentrations are modest in the context of the Cheshire and Shropshire meres, which are renowned for their high TP levels but low nitrate values (Moss *et al.*, 1994). The lake appears to be nitrogen rather than phosphorus limited because chlorophyll a concentrations are on average not high (e.g. annual mean of 11 μg l^{-1}).

In summary, Betton Pool has experienced eutrophication over the last few decades, resulting in a change from a naturally eutrophic lake to a hypertrophic one in terms of TP concentrations (> 100 μg TP l^{-1}). Moss *et al.* (1992) concluded that the lake did not appear to be suffering from eutrophication problems that were any greater than the probable regional change that had occurred in the post-war period, as a result of agricultural intensification. However, continued monitoring of nutrient concentrations in the lake would be desirable based on the palaeolimnological findings of the current study.

Figure 44 Lithostratigraphic data for Betton Pool

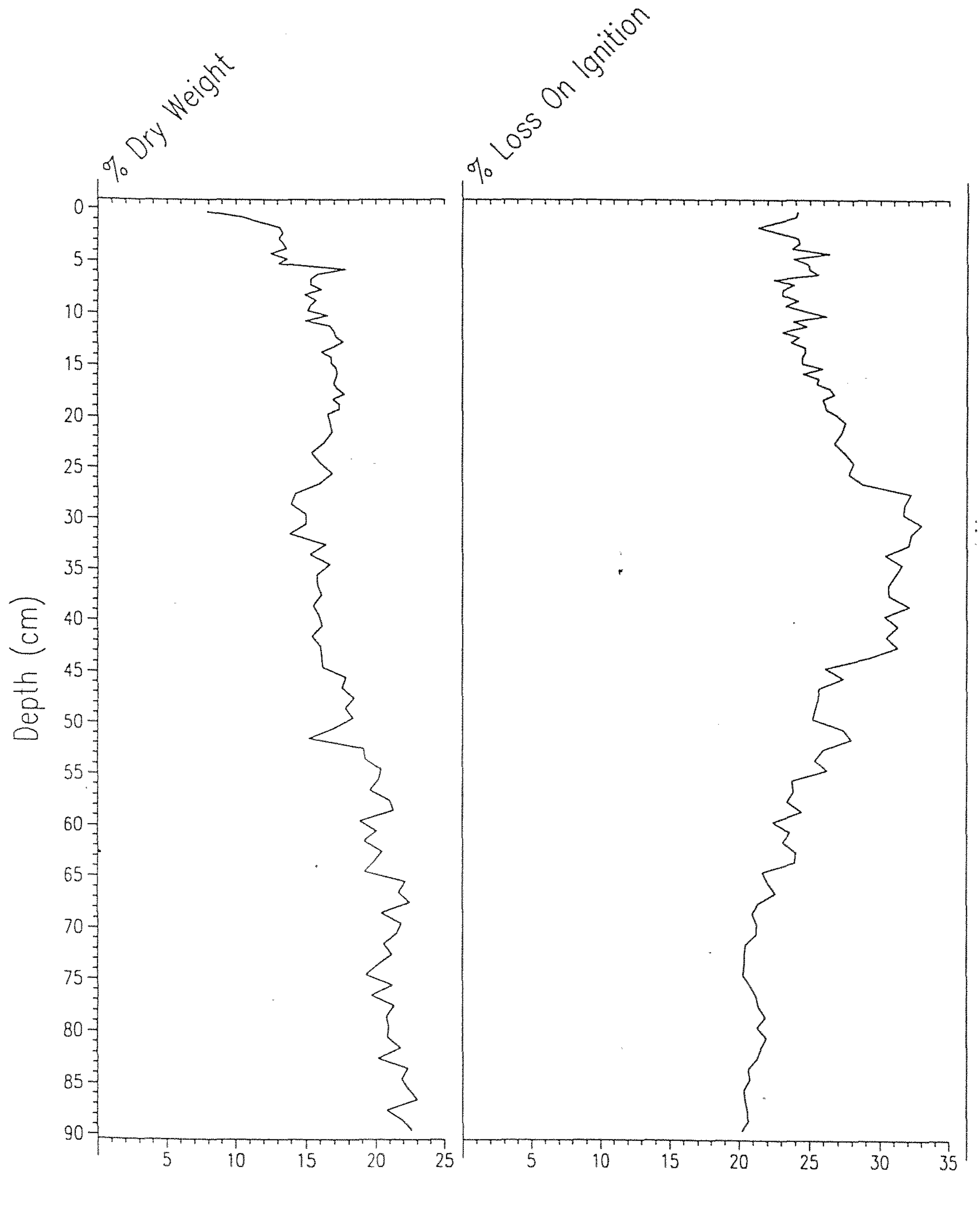


Figure 45 ^{210}Pb Activity versus Depth - Betton Pool

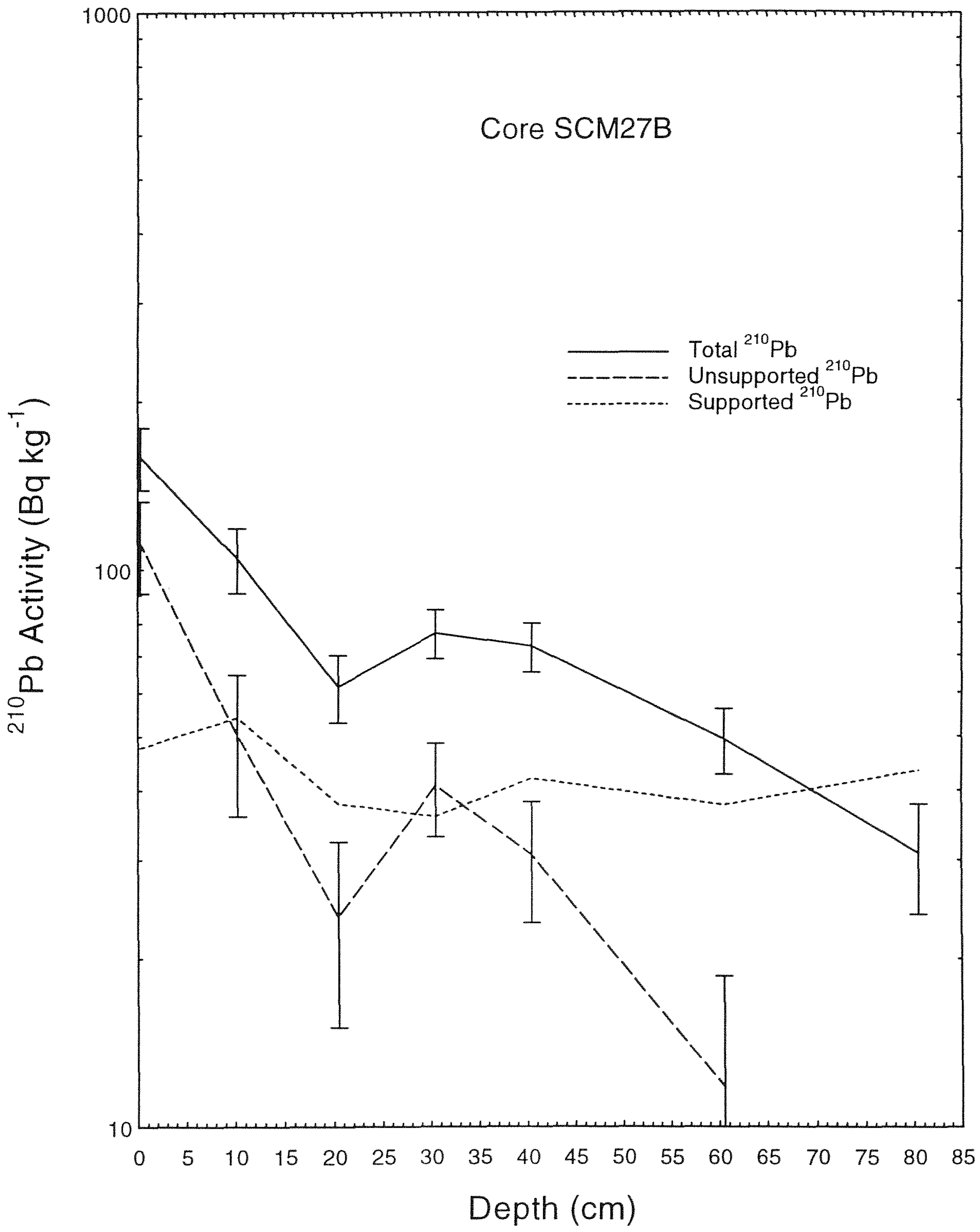
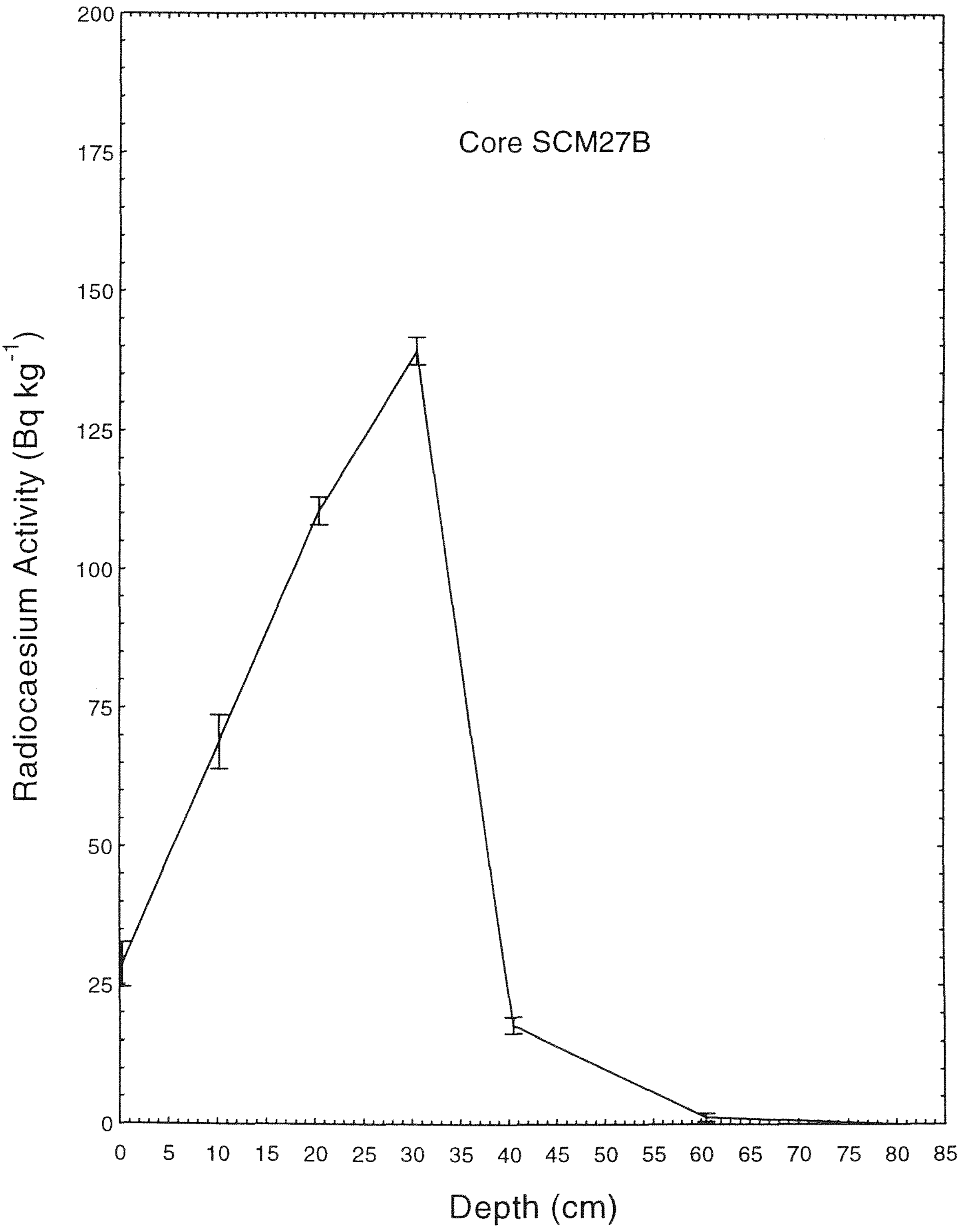


Figure 46 ^{137}Cs Activity versus Depth - Betton Pool



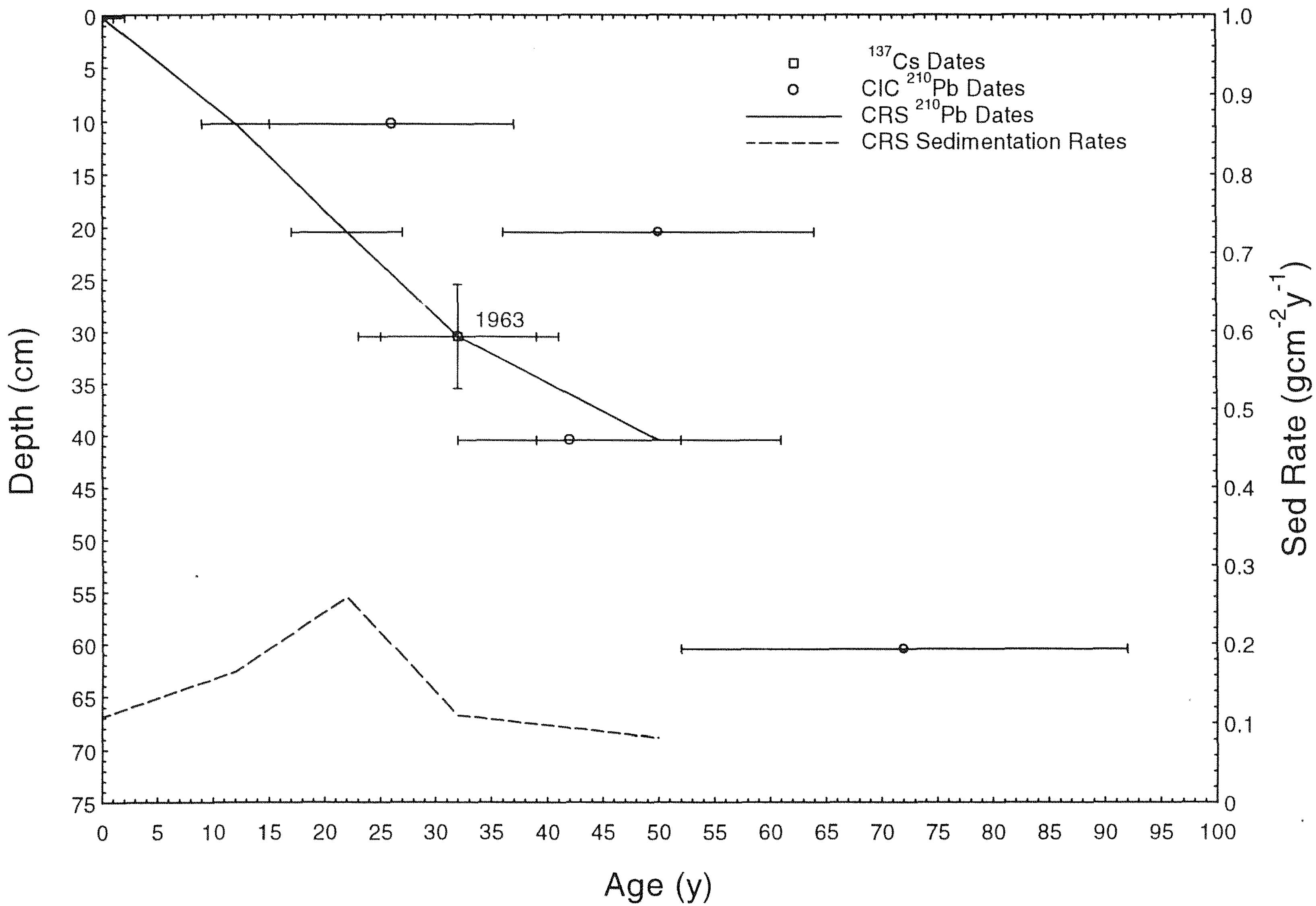
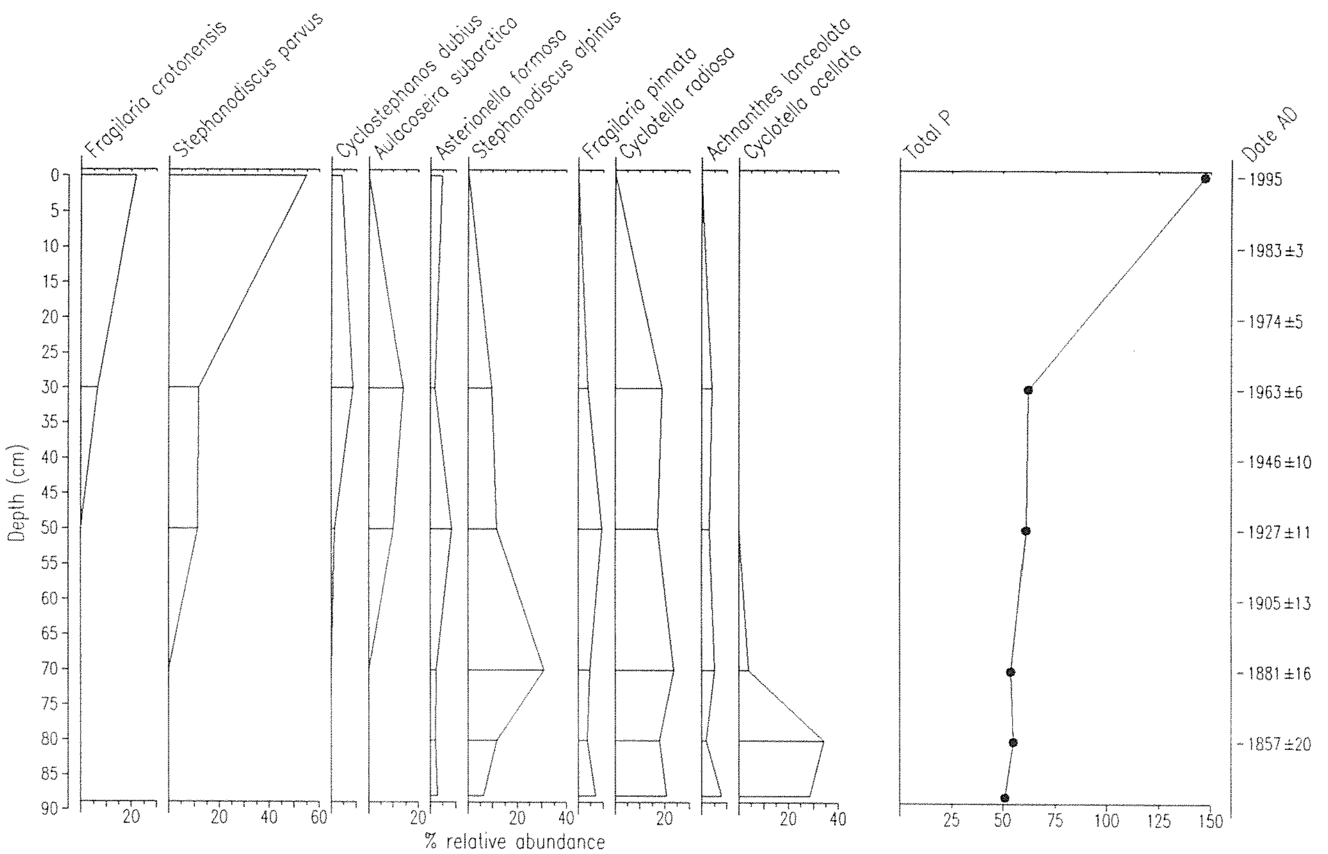


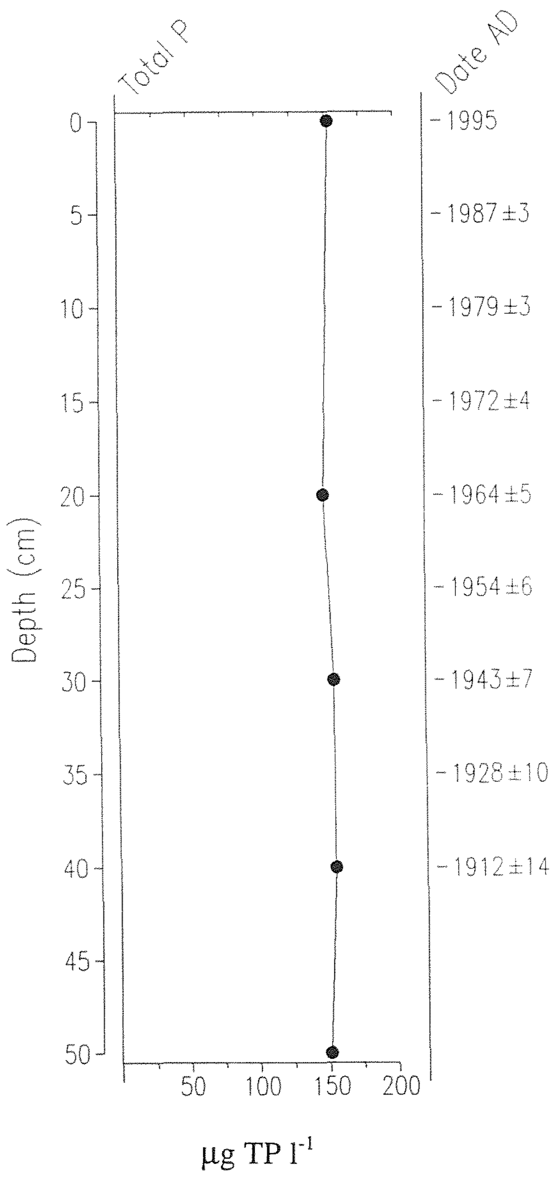
Figure 47 Depth versus Age - Betton Pool

Figure 48 Summary diatom diagram and reconstruction for Betton Pool



Upton Broad

- **The DI-TP results indicates that Upton Broad is a typical, shallow, nutrient-rich lake and has not experienced any marked changes in TP concentrations over the last century.**
- **The diatom assemblages are dominated by *Fragilaria* taxa, commonly observed in such waters but the lack of planktonic taxa causes some uncertainty over the diatom model results.**



12. Upton Broad, Norfolk (TG 388 134)

12.1 Site Description and Water Chemistry

- Altitude 0 m
- Area 5.2 ha
- Lies in the Bure valley, separated from the river by a sluice
- Mean depth 1.3 m
- Maximum depth 1.7 m
- A spring-fed broad
- The shore substrate consists of peat and muds
- Surrounded completely by open reed fen with scrub and alder carr in places.
- The catchment is half carr and half arable land
- Unique amongst the Broads because of its unusual sediments which are largely composed of invertebrate faecal pellets containing preserved algal cells, and its exceptionally clear water.

pH	units	8.7
conductivity	$\mu S\ cm^{-1}$	559
alkalinity	mg $CaCO_3$	116
calcium	mg l^{-1}	72.5
total nitrogen	mg l^{-1}	1.4
total phosphorus	$\mu g\ l^{-1}$	38

12.2 Macrophyte Survey

Date of visit 28-7-96.
A map of aquatic macrophyte distribution for Upton Broad is presented in Figure 49. The site is shallow, with a maximum depth of approximately 2.0 m. Water transparency at the time of survey was good and it was possible to view the lake bottom throughout. The surface substrate is almost exclusively composed of soft, green, organic mud which is largely composed of the faecal pellets of invertebrates, and appears to be very unstable and very possibly prone to resuspension during stormy weather. The aquatic macroflora is generally typical of the more undisturbed Broadland sites in Norfolk although the submerged flora is relatively species poor. Upton Broad is ringed by *Phragmites australis* dominated fen in which *Cladium mariscus* is also abundant. *Typha angustifolia* is also locally abundant in some locations and common associated fen taxa include *Iris pseudacorus*, *Oenanthe crocata*, *Lycopus palustris*, *Caltha palustris*, *Carex* spp. and *Mentha aquatica*.

The most abundant submerged macrophyte is *Najas marina* which grows throughout the lake to a water depth of approximately 1.5 m and in places forms dense stands. Ratcliffe (1977) highlighted Upton Broad as the main site for this species, for which the Broads are the only British locality. *Potamogeton friesii* was found in an isolated location in the south-east of the site, while small quantities of *Zannichellia palustris* were retrieved from close to the shoreline in the south-west corner. *Nymphaea alba* occurs in significant stands, particularly in the east of the site, and in the north-east corner forms an open association with *T. angustifolia*. The charophyte *Chara vulgaris* var. *vulgaris* is present in small quantities in a number of locations.

The aquatic macrophyte composition of Upton Broad has been regularly monitored by the Broads Authority as part of the Norfolk Broads macrophyte monitoring programme during the last few years. The submerged and floating leaved species list produced in their 1993 report (Kennison and Prigmore, 1994) is very similar to that produced by this survey. However, in 1993, *Potamogeton pectinatus* occurred in several locations and no charophytes were found (as was the case for 1992, despite previous records). Ratcliffe (1977) refers to *Nymphaea alba* and *Nuphar lutea* as being locally co-dominant, however there has been no recent record of the latter species at this site. These apparent differences are unlikely to be signify major environmental change at this site, although the reappearance of charophytes after at least two years of absence might reflect a return to better water quality after a short period of deterioration.

In its evaluation of the 1993 macrophyte data, the Broads Authority classified Upton Broad as belonging to the Phase I to Phase II group, which describes sites which are species rich and generally little impacted by increased nutrient levels. Phase III and Phase IV sites are those in which the macroflora has undergone increasing species impoverishment and change as a result of nutrient enrichment. The current survey would concur with this classification. Applying the species list presented in Table 13, Upton Broad is classified as Type 10B (Palmer, 1992), a eutrophic category.

Table 13 Species abundance and DAFOR abundance rating for Upton Broad

	abundance	notes
Submerged taxa		
Chara vulgaris var. *vulgaris*	O	isolated plants in scattered locations
Najas marina	A	locally dominant, throughout to a depth of 1.5 m
Potamogeton friesii	R	one specimen found
Zannichellia palustris	O	in s-e
Floating leaved taxa		
Nymphaea alba	F	substantial stands in east
Emergent taxa		
Caltha palustris	F	
Carex spp.	F	
Cladium mariscus	A	locally dominant
Epilobium spp.	F	
Iris pseudacorus	O	
Mentha aquatica	F	
Oenanthe crocata	O	
Phragmites australis	D	in all emergent habitats
Salix sp.	O	
Typha angustifolia	O	locally abundant

12.3 Lithostratigraphy

A 135 cm sediment core (UPTO1) was taken from the deepest part of the lake at a water depth of 1.6 m using a piston corer on 30-1-95. The lower part of the core (> 60 cm) was a dark olive brown (2.5Y 3/3) detrital lake mud with traces of calcareous material (Ld3 Dg1 Lc++ Dh+). Above 60 cm the sediment changed to an olive grey (5Y 4/2), although sediment composition remained largely unchanged. Diatoms were only well preserved above 40 cm. The upper few cms were a striking bright green colour, composed largely of the blue-green alga *Aphanotheca stagnina*. The surface mud was a faecal pellet ooze produced by chironomids in which algal cells were preserved free from decay. Upton Broad is known for its unusual sediments (Jackson, 1978).

The %dw and %loi profiles (Figure 50) show that the sediments of Upton Broad were similar throughout the core, with %dw fluctuating at c.10-15% and %loi at 25-35%. The %loi for the surface layer was 42% reflecting the high organic content of the faecal pellet ooze.

12.4 Radiometric Dating

This core appears to have a relatively simple ^{210}Pb profile. Sediment accumulation appears to be quite rapid, equilibrium with the supporting ^{226}Ra being reached at a depth of more than 50 cm. Calculations using the CRS and CIC ^{210}Pb dating models both indicate a more or less uniform sedimentation rate since c.1960 of 0.060±0.010 g cm^{-2}y^{-1} (0.64 cm y^{-1}), though there is some divergence in the deeper sections, possibly due to lower accumulation rates during the first few decades of this century.

There is however a significant contradiction between the ^{210}Pb and ^{137}Cs results. The ^{137}Cs measurements place the 1963 fallout peak at 10±2.5 cm. This suggests a mean post-1963 sedimentation rate of ~0.03 g $cm^{-2}y^{-1}$, just half the ^{210}Pb value. Since there is no evidence of mixing, the most likely cause of the discrepancy is post-depositional loss from the record. The inventories of both radionuclides are significantly below values expected from direct fallout, though the losses are however much greater for ^{137}Cs (~90%) than ^{210}Pb (~50%). In view of this, and the coarse resolution of ^{137}Cs peak, the ^{210}Pb dates are probably more secure. Table 14 gives CRS model ^{210}Pb dates, which are to some extent a compromise between the ^{137}Cs and CIC model. In light of the above discussion these should however be regarded with some caution. The results are illustrated in Figure 51, Figure 52 and Figure 53.

Table 14 Chronology of Upton Broad

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0	0.00	1995	0				
5	0.47	1987	8	3	0.060	0.64	19.1
10	0.94	1979	16	3	0.059	0.63	26.6
15	1.41	1972	23	4	0.061	0.63	27.8
20	1.87	1964	31	5	0.062	0.63	28.7
25	2.40	1954	41	6	0.052	0.51	30.9
30	2.93	1943	52	7	0.040	0.38	33.2
35	3.47	1928	67	10	0.034	0.31	36.7
40	4.00	1912	83	14	0.028	0.25	40.3

Extrapolated dates:

45 cm	1894
50 cm	1877
55 cm	1860

12.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in five levels of the sediment core were calculated and Figure 54 illustrates the results for the major taxa. Diatom preservation was good in the upper 20 cm and then became progressively poor downcore. Counting was not possible below 50 cm. The diatom assemblages had low diversity throughout the core and a total of only 20 taxa was observed, 16 of which were present in the calibration set. All of the common taxa were well represented in the calibration set, with greater than 94% of the fossil assemblage being used in the calibration procedure.

Figure 54 illustrates that there have been slight changes in the diatom species composition in terms of relative percentages over the period represented by the upper 50 cm of the core (c.1877-1995). However, there has been no clear species replacement and the assemblages have been dominated by the same non-planktonic *Fragilaria* taxa throughout the core, species commonly found attached to either the sediments or macrophyte surfaces of shallow, alkaline waters. The only changes were that *Fragilaria lapponica* and *Navicula graciloides* accounted for greater relative abundances in the 50 cm sample (c. 1877) than in the upper samples, *Fragilaria brevistriata* occurred in the highest frequencies in the 40 cm sample (c. 1912) and in the surface sample, and *Fragilaria construens* had the highest frequencies in the 30 cm sample (c. 1943), declining in importance towards the top of the core.

12.6 Total Phosphorus Reconstruction

The TP reconstruction shows that the lake has had high TP concentrations throughout the period represented by the sediment core (1877-1995), which would place the lake in the hypertrophic category (> 100 μg TP l^{-1}). The model indicates that there has been no change in the trophic status of the lake with values always c. 150 μg TP l^{-1}. The diatom-inferred concentrations were 151 μg TP l^{-1} for c. 1877 (50 cm), 155 μg TP l^{-1} for c. 1912 (40 cm) and c.1943 (30 cm), 148 μg TP l^{-1} for c.1964 (20 cm), and 153 μg TP l^{-1} for 1995.

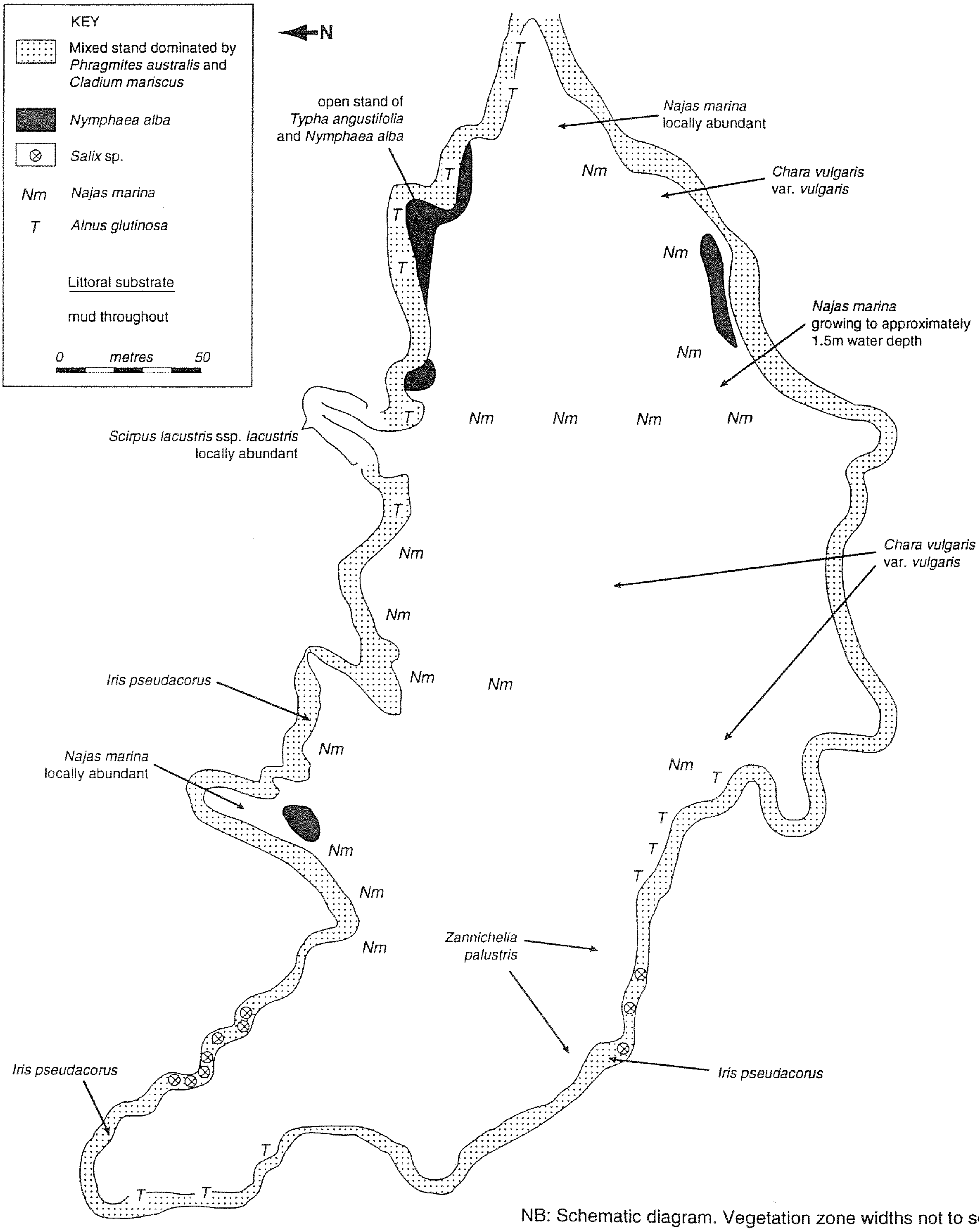
12.7 Discussion

The model results suggest that Upton Broad is a naturally eutrophic lake and has not experienced any major increase in TP concentrations since 1877. However, these data must be viewed with caution in view of the low diversity of the diatom assemblage and the absence of planktonic taxa. The lack of planktonic forms in the sediments causes uncertainty over the results of the model because the relationship between non-planktonic taxa and open water nutrient concentrations is not as direct as that for planktonic forms (Bennion, 1995). The absence of such forms, particularly the centric diatoms, may be related to the need for motility on the sediments because these are non-motile taxa and would be buried in the deposits whenever there was disturbance by wave action etc and would be unlikely to survive (Round, 1953). High grazing pressure by zooplankton may also suppress the phytoplankton. The dominance of the benthic *Fragilaria* spp. could be because of the clear water conditions in Upton Broad which allow light to penetrate to the surface sediment so that benthic forms can grow there *in situ.* Likewise, the presence of epiphytic taxa in the diatom assemblages reflects the availability of submerged macrophytes to act as hosts.

These factors may explain why the model over-estimates TP values for Upton Broad. In a 14 year water chemistry survey (1981-1994) conducted by the National Rivers Authority, annual mean TP concentrations ranged from 29-67 μg TP l^{-1} (Doarkes, pers. comm) whereas the model estimated TP values for the lake of c.150 μg TP l^{-1}. Similarly, in the 1995/6 survey by EA, TP ranged from 17-65 μg TP l^{-1}, thus the DI-TP is well outside this range. In fact, compared with other Broads, the nutrient concentrations in Upton Broad are low and the water is very clear (eg. Bennion, 1996). It is thought that the site represents the type of condition which existed in many other Broads prior to the decline in submerged vegetation (eg. Jackson, 1978; cf. Bennion, 1996). In a 1993 aquatic macrophyte survey (Kennison & Prigmore, 1994), Upton Broad was classed as a "Phase 1 to phase 2 broad: macrophytes abundant and species-rich in most places".Unlike many of the other Broads, there is no distinct inlet and the lake is surrounded by extensive woodland, acting as a buffer to prevent the inflow of agricultural fertilisers.

In summary, the diatom results indicate that Upton Broad is a typical eutrophic lake (although measured TP concentrations suggest that the lake is mesotrophic) and has not experienced any marked changes in TP concentrations over the last 100 years. The diatom flora is stable throughout the core and is dominated by *Fragilaria* taxa, commonly observed in clear, shallow, alkaline, nutrient-rich waters (Bennion, 1994; 1995).

Figure 49 Aquatic macrophyte distribution map for Upton Broad



NB: Schematic diagram. Vegetation zone widths not to scale

Figure 50 Lithostratigraphic data for Upton Broad

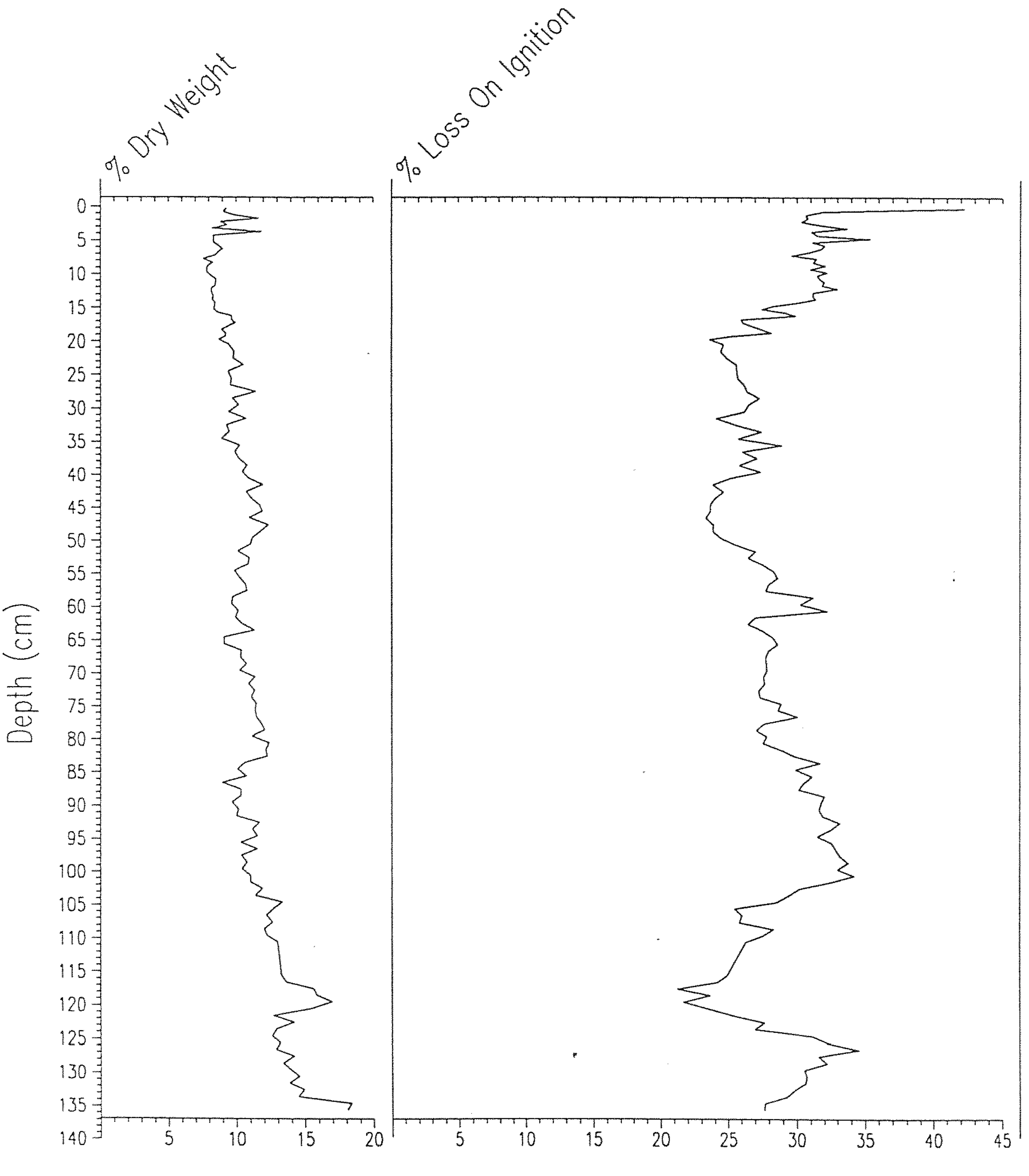


Figure 51 ^{210}Pb Activity versus Depth - Upton Broad

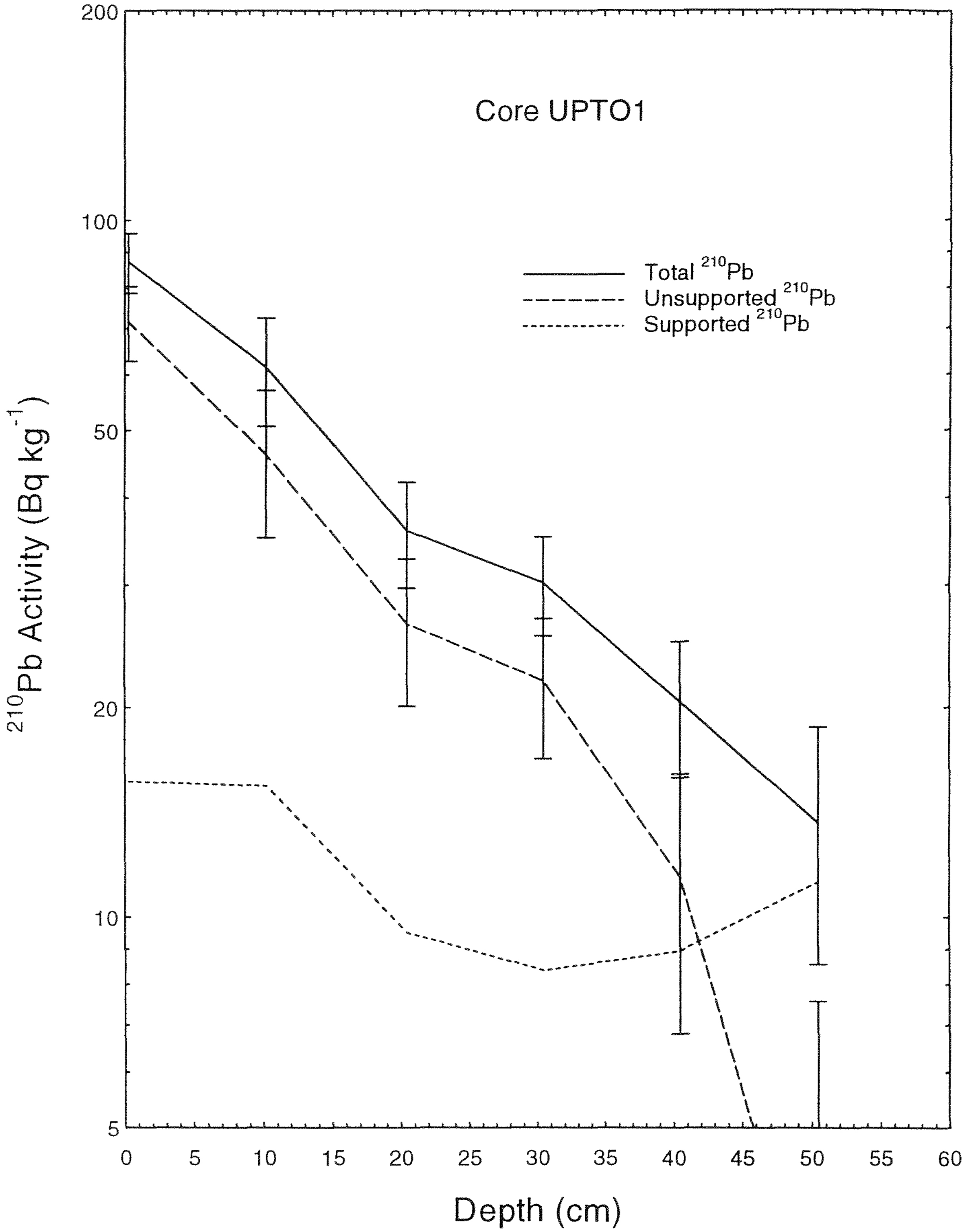


Figure 52 ^{137}Cs Activity versus Depth - Upton Broad

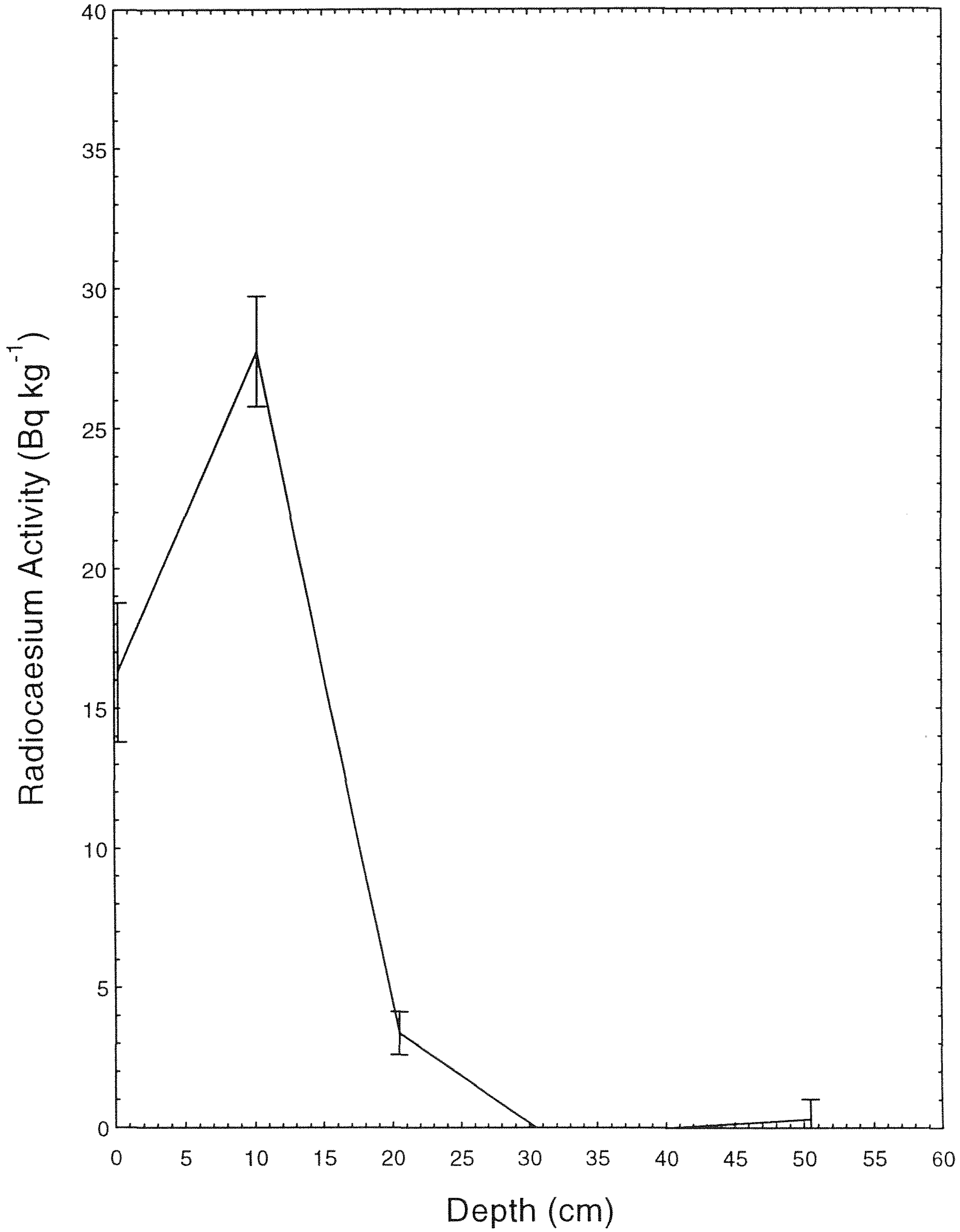


Figure 53 Depth versus Age - Upton Broad

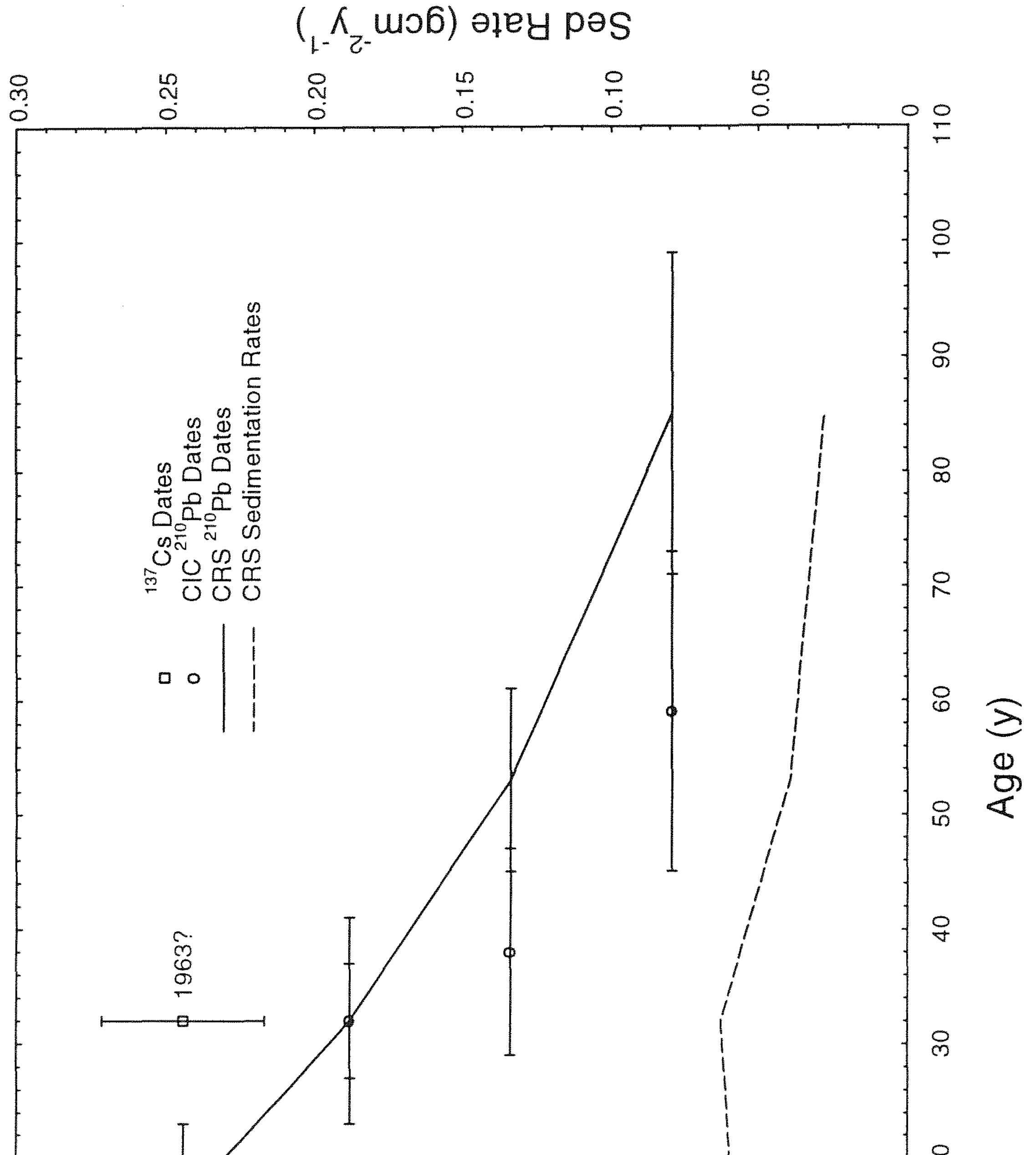
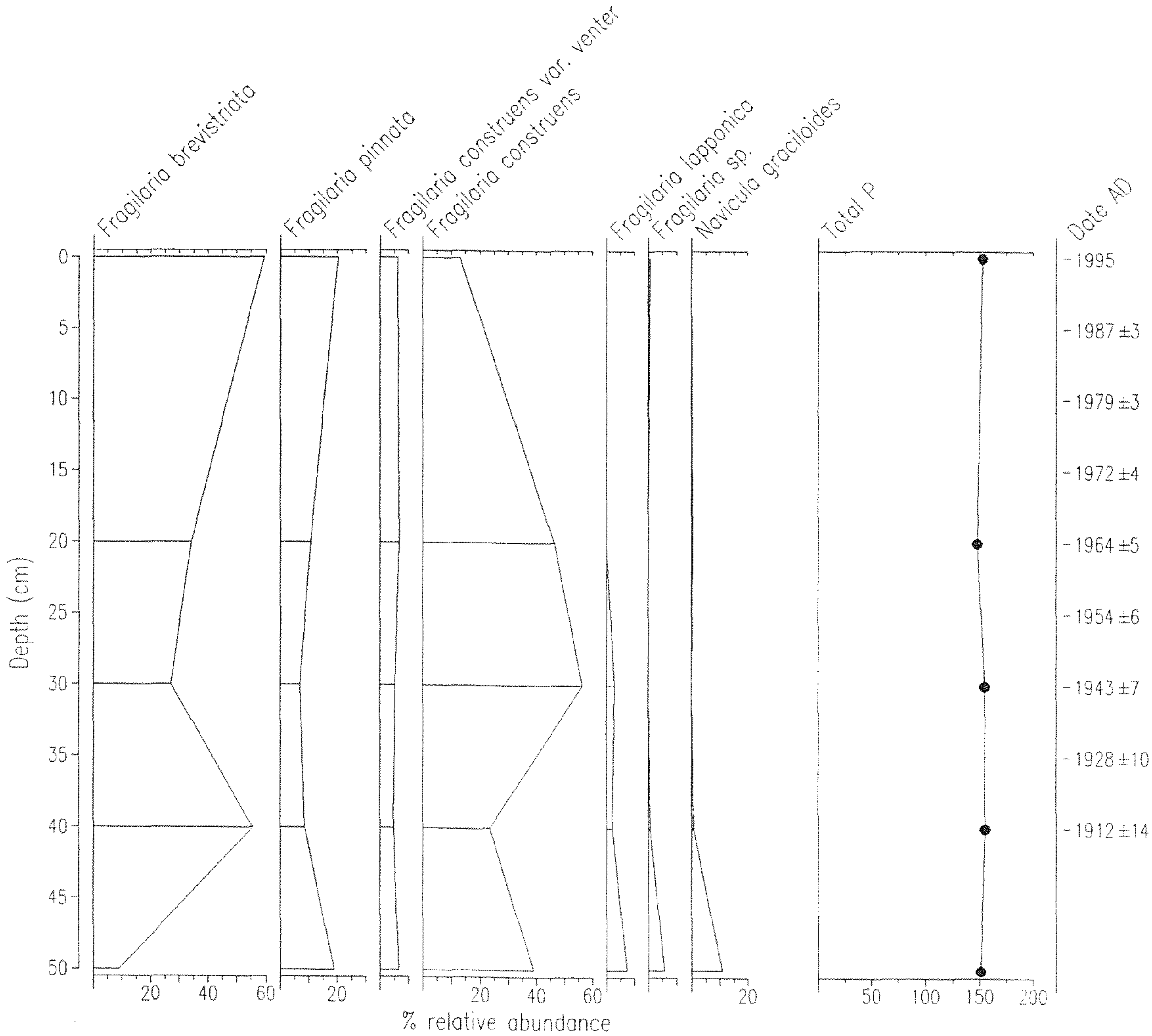
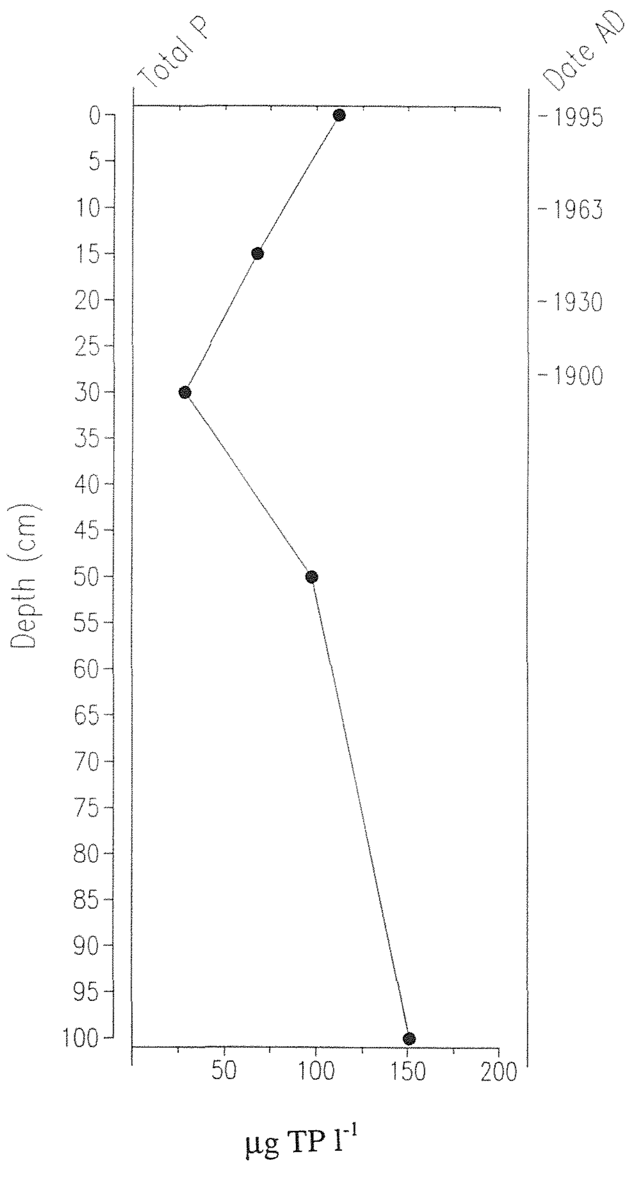


Figure 54 Summary diatom diagram and reconstruction for Upton Broad



Martham South Broad

- **The TP diatom model cannot be used with any confidence to reconstruct the nutrient history of Martham South Broad, owing to species analogue problems.**
- **The diatom flora, however, does suggest that the lake has always been brackish.**
- **There have been species shifts over the last century or so which appear to indicate fluctuations in salinity rather than any change in nutrient status.**



13. Martham South Broad, Norfolk (TG 458 201)

13.1 Site Description and Water Chemistry

- Martham South Broad is situated in the River Thurne system of the Norfolk Broads, a series of small lakes formed by flooding of peat excavations around the fourteenth and fifteenth centuries.
- The broad is brackish and the salinity is maintained by percolation of seawater under the coastal dunes bordering the North Sea
- Mean depth c. 1.0 m
- Maximum depth 1.5 m
- The land use is mostly rough grassland with some arable

pH	units	8.2
conductivity	$\mu S\ cm^{-1}$	6090
alkalinity	mg $CaCO_3$	149
calcium	mg l^{-1}	316
total nitrogen	mg l^{-1}	1.1
total phosphorus	$\mu g\ l^{-1}$	31

13.2 Macrophyte Survey

Date of visit 19-9-95.
A species distribution map for the site is presented in Figure 55. Water clarity was very good and as the site is shallow (maximum depth 1.2m) it was possible to acertain the zonation of the major stands of submerged macrophytes throughout the site without the use of a underwater viewing apparatus. However, an Ekman grab and double-headed rake were used to retrieve plants for identification.

The broad is mostly fringed by *Phragmites australis* and *Cladium mariscus* dominated reed-swamp, although part of the shoreline on the south side was open deciduous woodland. Open water across the south-eastern half of the broad is dominated by charophyte beds composed of several *Chara* ssp., including *Chara hispida*, *Chara apera* var *aspera* and *Chara vulgaris* var. *vulgaris* The south-western end is dominated by a substantial bed of *Najas marina*, which also occurs in lesser abundance off the north shore where it is flanked by a stand of *Hippuris vulgaris.* Specimens of *Nitellopsis obtusa* were recovered from within the main *Najas* bed.

The two bays in the north-east contain the more diverse macrophyte assemblages. A mixed stand of *Nuphar lutea* and *Nymphaea alba* occupies the arm connecting the water body to the North Broad, and *Callitriche* sp. is locally abundant under the floating leaf canopy. A single individual of *Potamogeton perfoliatus* was recovered from a rake trawl in the open water of the main bay. The *Chara* dominated south-eastern bay includes the occasional *Myriophyllum spicatum*, *Zannichellia palustris* and *Potamogeton pectinatus. P. pectinatus* is locally abundant in several locations adjacent to the eastern shoreline.

Martham South Broad has also been subject to regular monitoring by the Broads Authority in recent years. According to their 1993 report (Kennison and Prigmore, 1994), the Broad is placed in the category of Phase I to Phase II Broads (see section 12.2) and the current survey concurs with this and with the species list produced in the 1993 report. The site exhibits a diverse aquatic macroflora and there is no evidence to suggest significant environmental change over the last few decades. Application of the site typing scheme of Palmer (1992) to the species list in Table 15, types Martham South Broad as type 10, a eutrophic category.

Table 15 Species list and DAFOR abundance rating for Martham South Broad

	abundance	notes
Submerged taxa		
Chara hispida	F	locally dominant in s-e
Chara aspera	O	most frequent in s-e
Chara vulgaris	O	most frequent in s-e
Nitellopsis obtusa	R	within *Najas* stand
Najas marina	F	locally abundant
Hippuris vulgaris	O	locally abundant
Potamogeton perfoliatus	R	single specimen found
Potamogeton pectinatus	O	locally abundant in east
Zannichellia palustris	O	close to east shore
Myriophyllum spicatum	O	close to east shore
Elodea canadensis	R	in north-east
Floating leaved taxa		
Nuphar lutea	R	locally abundant in north-east
Nymphaea alba	R	locally abundant in north-east
Emergent taxa		
Phragmites australis	A	locally dominant
Cladium mariscus	A	locally dominant
Typha angustifolia		

13.3 Lithostratigraphy

A 104 cm sediment core (MART1) was taken in a water depth of 1.0 m using a piston corer on 19-9-95. The lower part of the sediment core (> 60 cm) was a very dark brown (10YR 2/2) lake mud with some silt and plant remains (Ld3 Ag1 Lso+ Lc+ Dh+).The upper 60 cms were a very dark grey (10YR 3/1) inorganic, calcareous sediment with small mollusc shells (Ld2 Lc2 Lso+ Ag+).

The %dw and %loi profiles (Figure 56) revealed a clear change at c. 60 cm from a more organic sediment with %loi values of c. 25-30% and %dw of around 20-25%, to less organic, calcareous material with %loi values of 5-10% and %dw of around 25-30% in the upper part of the core. The wd data followed the same pattern with generally higher values in the upper core section.

13.4 Radiometric Dating

Very low unsupported ^{210}Pb activities in this core recorded throughout the top 30 cm do not allow dating by the ^{210}Pb method. The ^{210}Pb inventory in the top 30 cm was just c.500 Bq m^{-2}, less than 30% of the fallout record (Figure 57).

The ^{137}Cs profile has a well defined maximum at 10.25 cm, that almost certainly records the 1963 weapons fallout peak (Figure 58). This suggests an accumulation rate of c.0.3 cm y^{-1}, which would put 1930 at c.20 cm, and 1900 at c.28 cm.

13.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in five levels of the sediment core were calculated and Figure 59 illustrates the results for the major taxa. Diatom preservation was satisfactory throughout the core. A total of 90 taxa was observed, 64 of which were present in the calibration set. There were analogue problems throughout the core owing to the relative importance of brackish diatom species in the fossil assemblages that were not present in the calibration set. Analogues were particularly poor for the 30 cm sample and to a lesser extent the 100 cm sample because of the high percentage of *Mastogloia smithii*, a brackish species which is absent from the Northwest European freshwater dataset. In view of these analogue problems, the TP reconstruction should be interpreted with caution.

Figure 59 illustrates that there have been marked changes in the diatom species composition over the period represented by the 100cm core. The radiometric dating results indicate that the core represents at least the post-1800 period. The bottom sample (100 cm) was dominated by the non-planktonic *Fragilaria* spp., typical of shallow, alkaline waters, with relatively high abundances of *Cocconeis placentula* and *Rhoicosphenia curvata*, species commonly found attached to plants in high conductivity waters. There was no major difference between the 100 cm and the 50 cm sample (c.1850) except for the appearance of *Gomphonema angustatum* and disappearance of *Rhoicosphenia curvata.*

The assemblage at 30 cm (c.1900) was, however, slightly different from the lower levels with high percentages of *Achnanthes minutissima* and very low abundances of the *Fragilaria* spp. *Cymbella microcephala* increased in importance and *Mastogloia smithii*, a brackish species, constituted 20% of the diatom assemblage. There was a further change by the 15 cm sample (c.1950). This sample was dominated by *Gomphonema angustatum,* and *Achnanthes minutissima* was also important. The *Fragilaria* spp. dominated the surface sample, although brackish species, especially *Amphora coffeaeformis,* were still present.

13.6 Total Phosphorus Reconstruction

The TP reconstruction shows that there have been changes in the nutrient status of Martham South Broad. Given the analogue problems, however, the diatom-inferred results are not reliable and only a qualitative interpretation of the species changes can be made. The values produced by the model are clearly erratic ranging from 150 $\mu g\ l^{-1}$ for the bottom sample to only 28 $\mu g\ l^{-1}$ for the 30 cm sample, increasing again to 112 μg TP l^{-1} for the 1995 sample.

13.7 Discussion

Monitored TP data over the period 1982-1992 gave an average TP concentration of c. 40 μg TP

l^{-1} and showed that there has been no marked change in TP concentrations over the 10 year period (unpublished). The 1995/6 water quality survey by the EA recorded TP values ranging from 5 to 80 μg TP l^{-1} with a data set mean of 31 μg TP l^{-1}. Therefore, there is a clear discrepancy between the measured and modelled data. It is likely that the species shifts in the core are a response to changes in lake conductivity rather than nutrient concentrations because salinity is maintained by percolation under the coastal dunes bordering the North Sea. Hence, chloride concentrations are high, ranging from 1000 to 2000 mg l^{-1} in the 1982-92 dataset. A number of the taxa in the fossil assemblages can tolerate highly conductive waters, e.g. *Cocconeis thumensis, Amphora coffeaeformis* and *Mastogloia smithii* and are therefore indicative of more saline periods in the lake's history, possibly due to sea water incursions.. *Mastogloia smithii* is also a major epiphyte in Hickling Broad, which is another broad with highly conductive waters (B. Moss, pers. comm.).

The Thurne system was originally estuarine and the Hundred stream discharged to the sea until approximately the eighteenth century. Martham South Broad would then have been entirely isolated from the stream or essentially marine in character. In contrast the *Fragilaria* spp. and *Achnanthes minutissima* are typical freshwater taxa, commonly observed in shallow lakes with either clear water and/or submerged macrophytes and represent less saline phases in the lake's history.

The lack of plankton in Martham South Broad, as for a number of the other shallow lakes sampled in this study, poses additional problems for inferring epilimnetic TP concentrations from diatoms. The dominance of such taxa may well be explained by extent of macrophyte growth or penetration of light rather than nutrient concentrations in the water column. Martham South Broad is one of the few remaining Phase 1/Phase 2 Broads (Kennison & Prigmore, 1994) because of its abundant macrophytes, particularly charophytes, and clear water conditions. Therefore, one would expect the diatom assemblages to be dominated by epiphytic forms, growing on the plant surfaces, and epipelic forms, living *in situ* on the sediment surface.

In summary, the diatom model cannot be used with any confidence to reconstruct the TP history of Martham South Broad, owing to species analogue problems. The diatom flora, however, does suggest that the lake has always been brackish. There have been species shifts over the period represented by the core which most likely indicate fluctuations in salinity rather than any change in nutrient status.

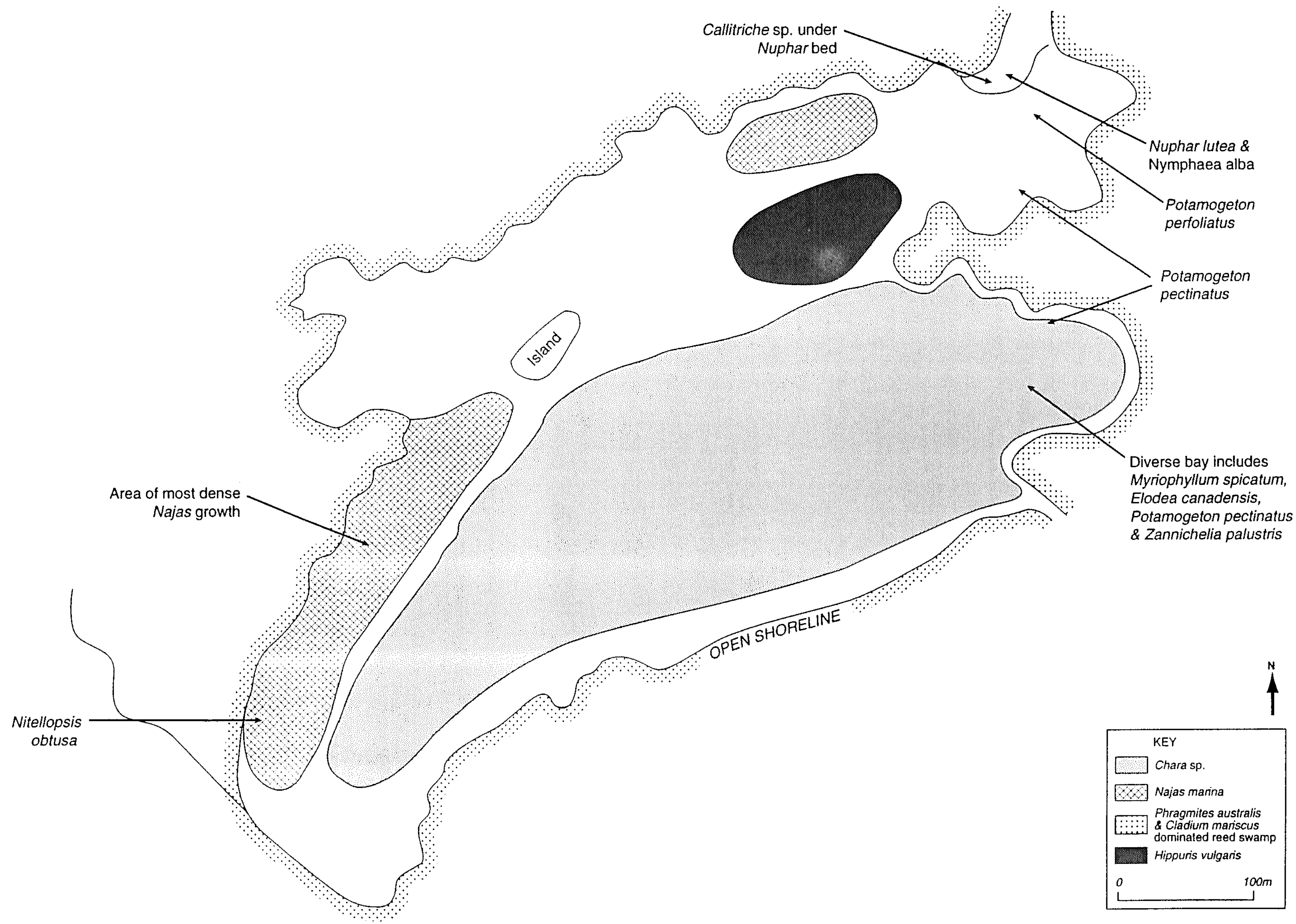


Figure 55 Aquatic macrophyte distribution map for Martham South Broad

Figure 56 Lithostratigraphic data for Martham South Broad

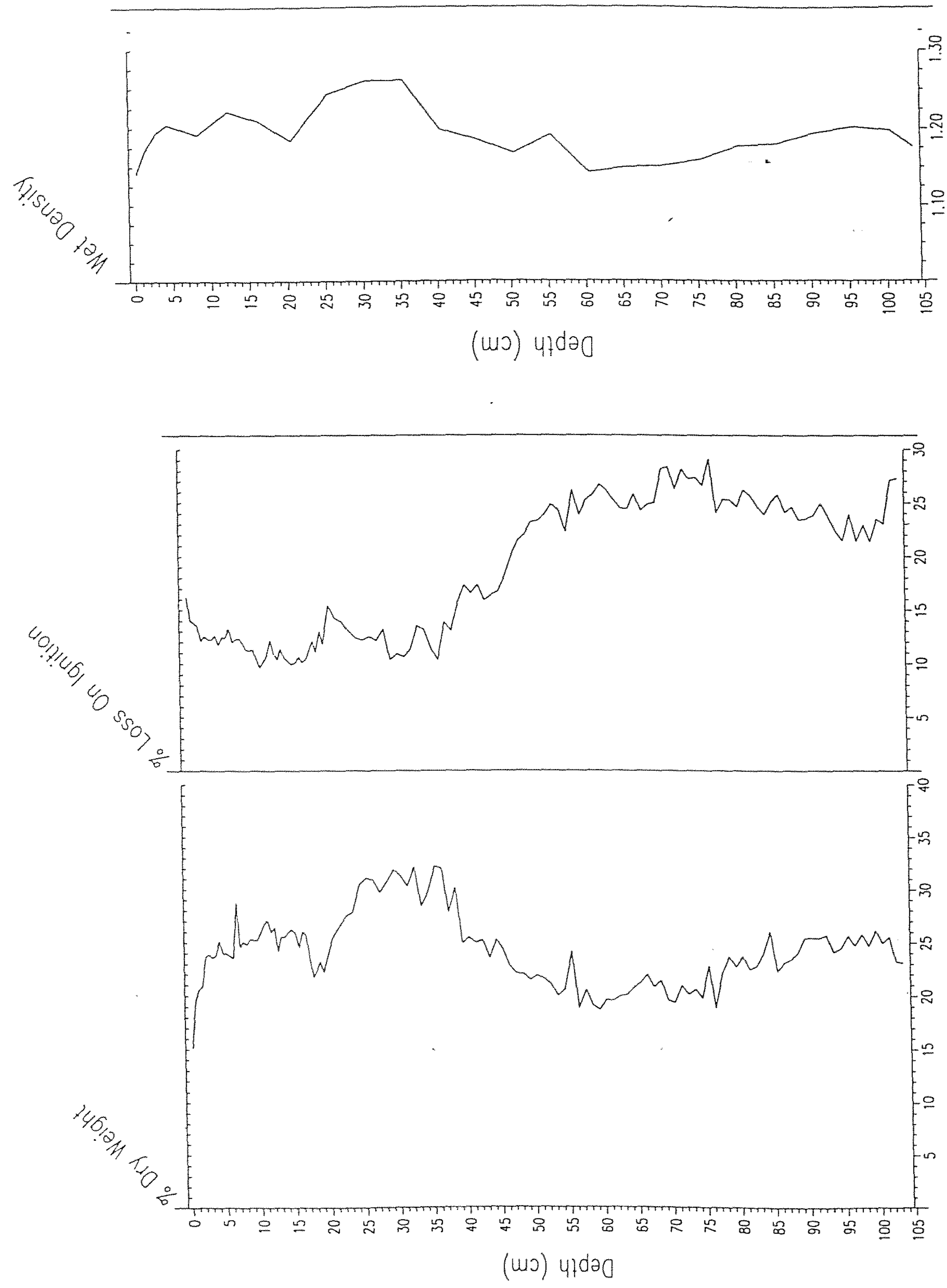


Figure 57 ^{210}Pb Activity versus Depth - Martham South Broad

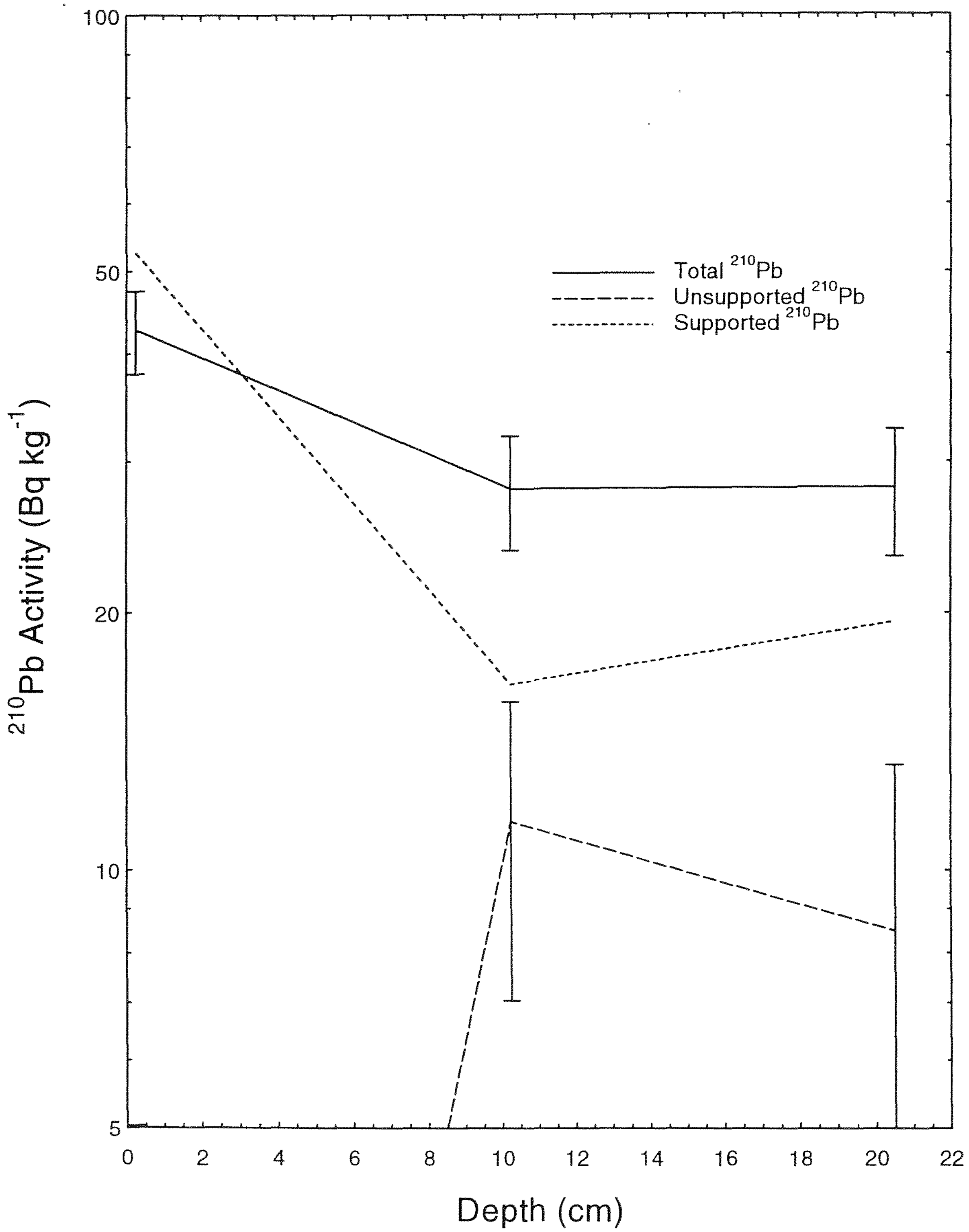


Figure 58 ^{137}Cs Activity versus Depth - Martham South Broad

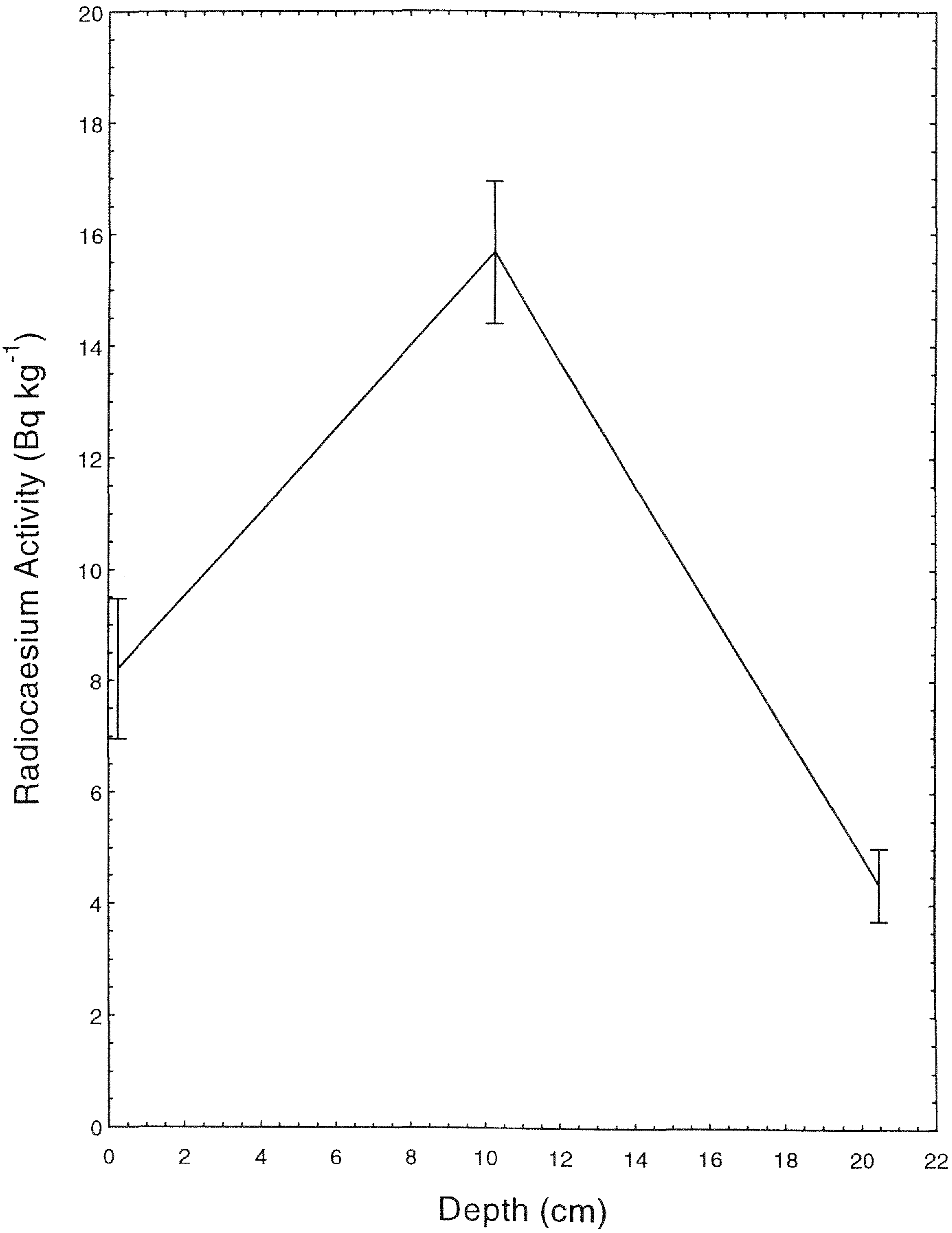
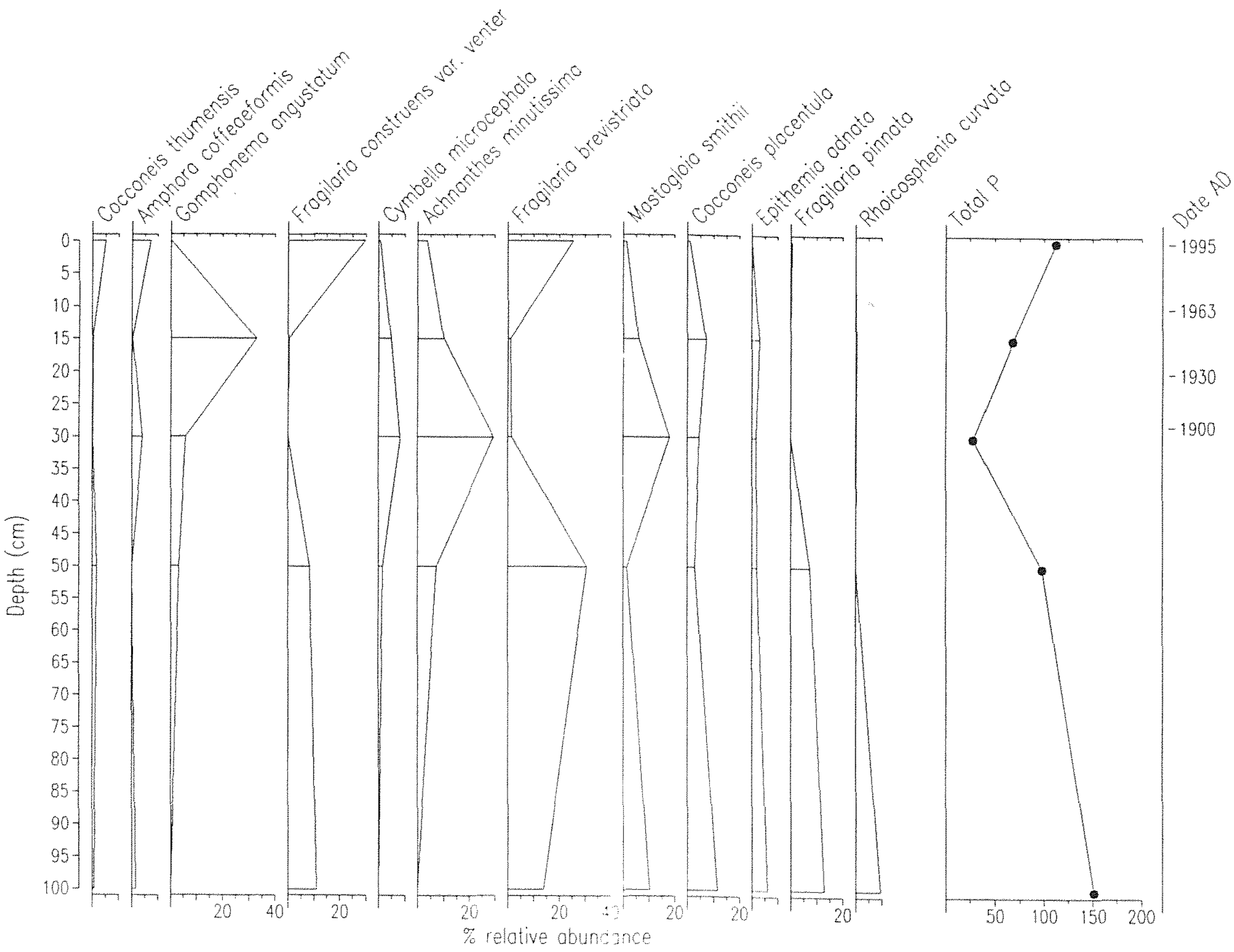
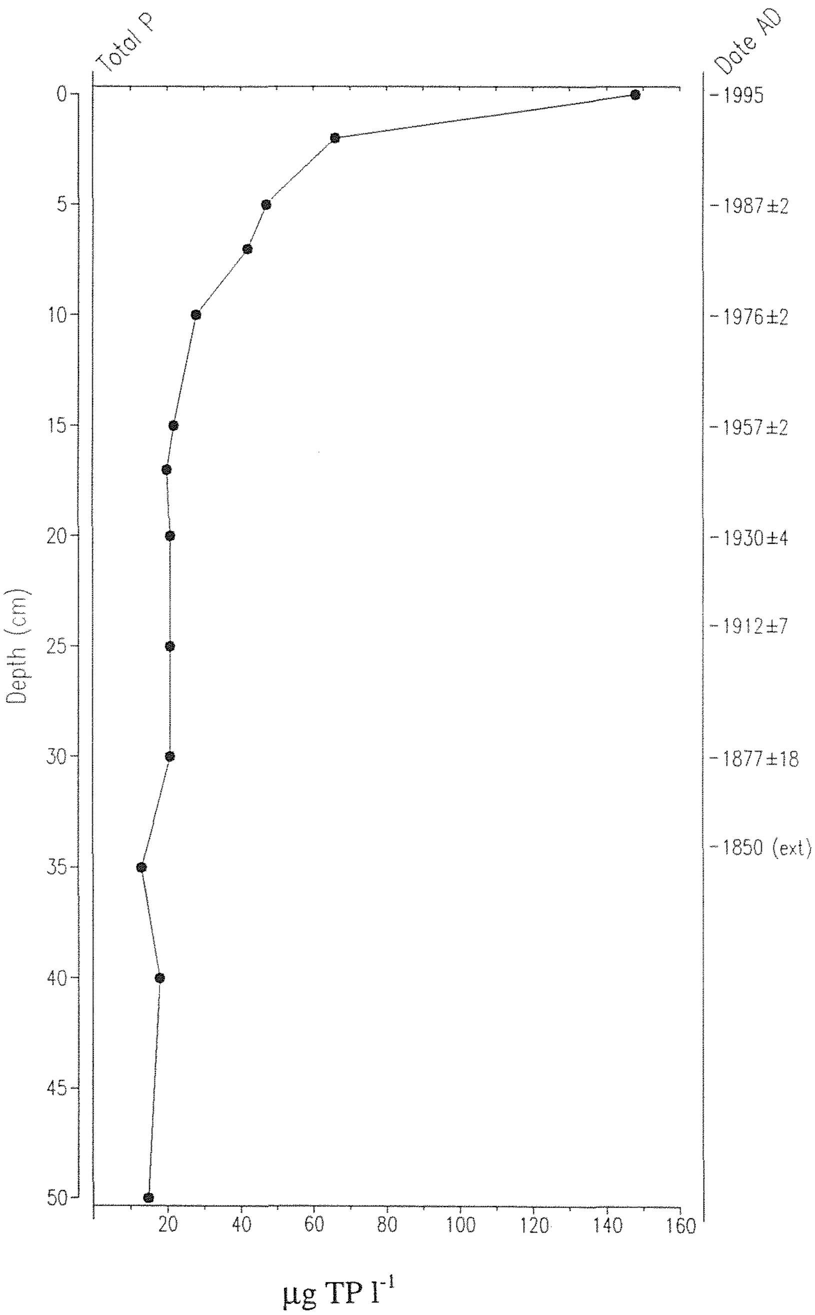


Figure 59 Summary diatom diagram and reconstruction for Martham South Broad



Esthwaite Water

- **Esthwaite Water has experienced eutrophication over the last two decades, switching from a naturally mesotrophic lake to a meso-eutrophic one.**
- **The lake is now dominated by planktonic diatom taxa typical of strongly enriched waters.**



14. Esthwaite Water, Cumbria (SD 358 969)

14.1 Site Description and Water Chemistry

- Esthwaite Water is one of the smallest and most productive lakes in the Cumbrian Lake District
- Altitude 65 m
- Area 1 km^2
- Mean depth 6.4 m
- Retention time of 13 weeks
- Maximum depth 15.5 m
- Approximately 45% of the total drainage area is cultivable; mostly lowland pasture

pH	units	7.5
conductivity	$\mu S\ cm^{-1}$	105
alkalinity	mg $CaCO_3$	23
calcium	mg l^{-1}	11.3
total nitrogen	mg l^{-1}	1.0
total phosphorus	$\mu g\ l^{-1}$	28

nb. These readings were taken from Cunsey Beck outflow

14.2 Macrophyte Survey

Date of visit 7/8-6-95.
A species distribution map and two transect profiles are presented in Figure 60 and Figure 61. The water at the time of the survey was turbid as a result of an algal bloom; secchi disc depth was 1.9m.

The west shore is characterised by stands of *Phragmites australis*, often fringing *Salix/Alnus* scrub, and muddy littoral sediments supporting occasional *Callitriche* sp., *Fontinalis antipyretica*, *Elodea canadensis* and *Nitella flexilis* var. *flexilis*. A single specimen of *Acorus calamus* was observed within an area of *Phragmites* swamp. The more exposed areas of shoreline on this side mostly comprise cobble and boulder substrates, with lawns of *Littorella uniflora* occurring in more sandy reaches. A transect in a north facing, sheltered bay (Transect 1) revealed a progression from: shoreline *P. australis* dominated reed bed; to *F. antipyretica* dominant in shallow water associated with *E. canadensis*, *L. uniflora*, *Callitriche* sp., *Nitella flexilis* var. *flexilis* and filamentous algae; to a mono-specific cover of *L.uniflora* between 0.6 - 0.9m; to occasional *Potamogeton obtusifolius* and *P. berchtoldii* in deeper water to the maximum depth of plant growth of 1.4m.

The inflow area at the north end is dominated by *Phragmites* reed beds, bordered by small stands of *Typha latifolia* and *Scirpus lacustris.* The open water is occupied by floating canopies of *Nuphar lutea* and *Nymphaea alba*; *F.antipyretica*, *P.obtusifolius* and *Lemna trisulca* being the most abundant submerged taxa.

The east shore is generally more exposed with coarse substrates ranging from sand and gravel to boulders. Few submerged or floating leaved species are evident although *L.uniflora* lawns are present on the finer substrates. The most common taxa are emergent/shoreline species such as *Polyganum hydropiper*, *Eleocharis palustris* and *Phalaris arundinacea*. However some sheltered bays contained stands of *Phragmites* with *Carex rostrata*, *Nuphar lutea* and *Nymphaea alba*, while *Nitella* sp,. and *F. antipyretica* are the dominant submerged taxa and *P.obtusifolius* is occasional. This association also dominates the south end although no submerged plants were found in the exceptionally turbid water close to the outflow.

Esthwaite Water has been the subject of several aquatic macrophyte surveys over the last century (Dr M. Wade pers. comm), the most recent of which was carried out by Newbold and Palmer in 1986 for the Nature Conservancy Council. There are several taxa which have been frequently recorded in the past but were not found in our survey. Among theses are certain species characteristic of oligotrophic waters, i.e. *Isoetes lacustris*, *Myriophyllum alterniflorum* and *Lobelia dortmanna*, which were recorded in 1986 and previously. It is possible that *I. lacustris* and *M. alterniflorum*, which usually have deeper water habitats, were overlooked, perhaps owing to the water turbidity at the time of the visit and the restricted use of the survey boat in open water due to strong winds. However, there was no evidence of shed *Isoetes* leaves in strand-line material. The absence of *L.dortmanna* is more difficult to account for given its preference for shallow water habitats. The possible loss of any of these species would be consistent with the chemical and palaeoecological evidence for a decline in water quality in recent times, however more survey work is required to verify this.

Najas flexilis, a nationally scarce species, for which there are records at this site, was not found during the 1986 or 1995 surveys (despite the use of snorkel divers in the former) and it is possible that it may also have been lost as a result of a deterioration in water quality at this site. However the absence of *Potamogeton crispus*, a species characteristic of more mesotrophic to eutrophic waters cannot be similarly accounted for and is more likely to have been overlooked.

Application of the species list presented in Table 16 types Esthwaite Water as type 5a (Palmer, 1992), a mesotrophic category.

Table 16 Species list and DAFOR abundance rating for Esthwaite Water

	abundance	notes
Submerged taxa		
Nitella flexilis var. *flexilis*	F	in south
Fontinalis antipyretica	A	widespread
Littorella uniflora	F	locally dominant
Potamogeton obtusifolius	F	in sheltered bays
Potamogeton berchtoldii	O	in south-east
Potamogeton pusillus	R	in north-west
Elodea canadensis	O	in west and north
Callitriche hamulata	O	widespread
Callitriche sp.	F	widespread
Sparganium minimum	O	in north and east
Lemna trisulca	R	abundant in north inflow bay
Ranunculus sp.	R	in north inflow bay
Floating leaved taxa		
Nuphar lutea	O	locally dominant in sheltered bays
Nymphaea alba	O	locally abundant
Emergent taxa		
Phragmites australis	A	dominant on south and west shores
Typha latifolia	R	locally abundant at north end
Scirpus lacustris var. *lacustris*	R	in north and north-west
Carex rostrata	F	locally dominant in sheltered bays
Eleocharis palustris	F	
Acorus calamus	R	single specimen on west shore
Alisma plantago-aquatica	R	on east shore
Phalaris arundinacea	F	widespread
Myosotis laxa	O	widespread
Caltha palustris	F	widespread
Polyganum hydropiper	O	on east shore
Agrostis stolonifera	O	widespread
Ranunculus flammula	O	widespread
Mentha aquatica	O	in west
Juncus effusus	F	widespread
Hydrocotyle vulgaris	O	widespread

14.3 Lithostratigraphy

An 86 cm sediment core (ESTH1) was taken from the northerly basin of the lake at a water depth of 15 m using a Mackereth corer on 7-6-95. The colour and sediment composition were similar throughout the core. The bottom sediments (25-80 cm) were a very dark brown (10YR 2/2) lake mud with traces of silt and plant fragments (Ld2 Lso1 Ag1 Dg+) and the upper 25 cms were a very dark greyish brown (2.5Y 3/2).

The %dw and %loi profiles (Figure 62) show that there has been little change with %dw c. 17% and %loi c.25% in the lower section of the core (40-86 cm). There was a slight and progressive increase in percentage organic matter from 35 cm to the core top, %dw falling to 5% and %loi rising to 30% at the sediment surface. Similarly there were no marked changes in wd with values of c. 1.0-1.1 g cm^{-3}

14.4 Radiometric Dating

Irregular ^{210}Pb activities above 15 cm suggest a rapid and sustained increase in sedimentation rates in recent decades following a long period of more uniform accumulation dating back to the second half of the 19th century. Mean accumulation rates during this earlier period were 0.037±0.002 g $cm^{-2}y^{-1}$, compared to contemporary values 2-3 times higher. Since dating of the top 15 cm by the CIC model was impractical, and there is no ambiguity in dates of the earlier levels, the results given below have been calculated using just the CRS model.

The ^{137}Cs measurements identified a major peak in activity at 5.25±2.5 cm depth recording fallout from the 1986 Chernobyl accident, corroborating the very recent CRS model ^{210}Pb dates which place the 1986 level at c.6 cm. Although the deeper ^{137}Cs measurements were insufficient to resolve the 1963 fallout peak with any accuracy, the ^{241}Am record suggests that this feature occurs at 11.5±2 cm, in reasonable agreement with the ^{210}Pb determined level of 14 cm.

The results are summarised in Table 17 and Figure 63, Figure 64 and Figure 65.

Table 17 Chronology of Esthwaite Water

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0.0	0.00	1995	0				
2.0	0.21	1993	2	2	0.100	0.85	7.2
4.0	0.48	1990	5	2	0.068	0.50	6.7
6.0	0.77	1985	10	2	0.065	0.44	8.0
8.0	1.06	1981	14	2	0.065	0.44	8.9
10.0	1.36	1976	19	2	0.059	0.39	8.2
12.0	1.67	1971	24	2	0.065	0.40	8.0
14.0	2.01	1963	32	2	0.044	0.26	8.5
16.0	2.36	1952	43	3	0.026	0.15	10.3
18.0	2.70	1941	54	4	0.041	0.23	13.8
20.0	3.06	1931	64	4	0.042	0.23	20.2
22.0	3.44	1922	73	6	0.043	0.23	23.2
24.0	3.82	1912	83	7	0.039	0.20	29.3
26.0	4.22	1902	93	10	0.036	0.17	38.7
28.0	4.64	1891	104	13	0.038	0.18	49.8
30.0	5.08	1877	118	18	0.027	0.12	59.6

Extrapolated Date:
1850 34 cm

14.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in thirteen levels of the sediment core were calculated and Figure 66 illustrates the results for the major taxa. Diatom preservation was good throughout the core. A total of 119 taxa was observed, 92 of which were present in the calibration set. All of the common taxa were well represented in the calibration set, with greater than 96% of the fossil assemblage being used in the calibration procedure.

Figure 66 illustrates that there has been a marked change in the diatom species composition since the 10 cm level, dated to 1976, following at least a 100 year period of relative stability. The eight lowermost samples representing the period c.1850 to c. 1976 were all dominated by mesotrophic taxa, particularly *Asterionella formosa*, *Aulacoseira subarctica*, *Achnanthes minutissima*, *Cyclotella comensis* and *Cyclotella radiosa*. However, there were slight changes in the relative abundance of these species over this period. *Cyclotella radiosa* and *Cyclotella comensis* decreased in relative importance whilst the relative frequency of *Asterionella formosa,* a spring diatom often observed in mesotrophic to eutrophic lakes increased. The first changes were the appearance of *Fragilaria crotonensis* and the rise in *Tabellaria flocculosa* at 20 cm (c. 1930), followed by

Stephanodiscus hantzschii, which was first observed in the 10 cm sample (c. 1976). These species changes indicate nutrient enrichment.

The uppermost five samples were markedly different from the lower samples. The mesotrophic taxa decreased in relative abundance, particularly *Cyclotella radiosa* and *Cyclotella comensis*, both of which disappeared. *Aulacoseira subarctica* and *T. flocculosa* also declined. Conversely, the relative abundances of *Asterionella formosa*, *Fragilaria crotonensis* and in particular *Stephanodiscus hantzschii* increased. *S. hantzschii* rose to 50% of the assemblage by 1995.

14.6 Total Phosphorus Reconstruction

The TP reconstruction indicates that the lake was mesotrophic until c.1980 with only small though steady increases in TP concentrations since c.1850. For example, the DI-TP concentrations were c. 15 μg TP l^{-1} at c.1850, increasing to c. 20 μg TP l^{-1} from c.1870-c.1960. Concentrations increased to 28 μg TP l^{-1} by c.1976 (10 cm), and then rose significantly and rapidly to c. 40 μg TP l^{-1} from 1985-1990, and then again to c. 65 μg TP l^{-1} by 1993. The model suggests that there has been a significant increase in TP concentration over the last few years with a DI-TP value for the surface sample of 150 μg TP l^{-1} related to the high percentage of *Stephanodiscus hantzschii*, a small, centric planktonic diatom commonly observed in highly enriched lakes. This long term stability followed by a recent increase in productivity is confirmed by the ^{210}Pb profile, which suggested a rapid and sustained increase in sediment accumulation rates in recent decades following a long period of more uniform accumulation dating back to the second half of the nineteenth century.

14.7 Discussion

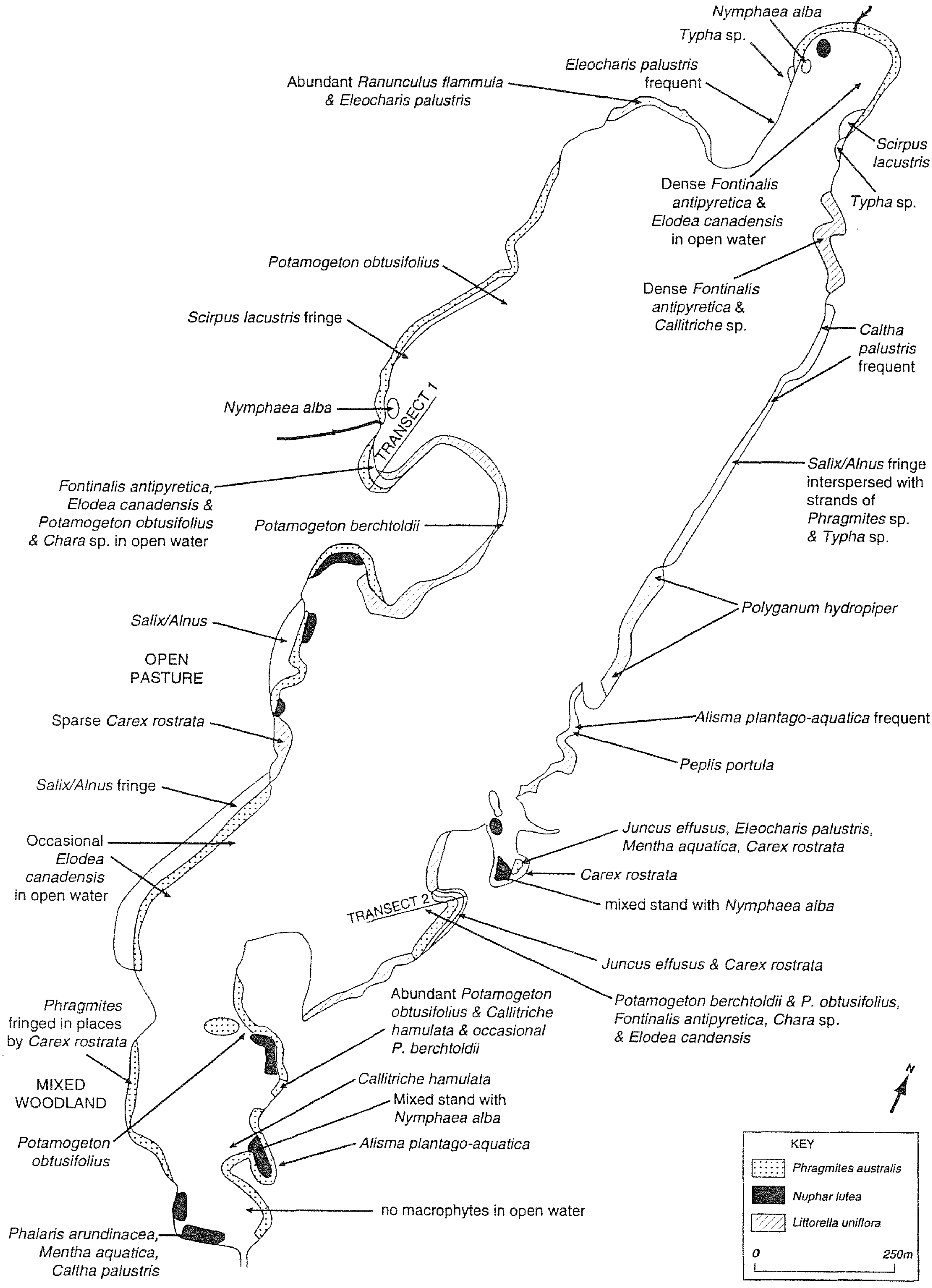
The significant recent eutrophication at Esthwaite Water, as inferred from the diatom model, is consistent with documented records of catchment events and water quality. The lake is considered to be the most productive of the larger lakes in the Lake District and was classed as a Group 3 silted productive lake with 45% of cultivable drainage area, large numbers of algae, and hypolimnetic anoxia in Pearsall's series (Gorham *et. al*, 1974). Nutrient concentrations and summer phytoplankton production have increased in the last few decades (Heaney *et al.*, 1986; Talling & Heaney, 1988; Heaney *et al.*, 1992), largely attributed to sewage inputs from Hawkshead sewage treatment works, opened in 1973. In 1986, a P removal system was installed but no improvement in water quality has been observed, largely due to internal P loading from the lake sediments and also perhaps due to diffuse sources of P from agriculture. In addition, a trout farm was established in 1981 and this may provide a further source of nutrients.

The inferred TP concentration of 28 μg TP l^{-1} for 1976 closely matches the historical water chemistry measurements of 25 μg TP l^{-1} in 1977 and 27 μg TP l^{-1} in 1986 (Talling & Heaney, 1988). There is a mis-match, however, between the diatom-inferred TP value for 1995 of 150 μg TP l^{-1} and the current measured TP of c. 30 μg TP l^{-1}, although a maximum TP concentration of 117 μg TP l^{-1} was recorded in 1995. This is because of the dominance of *Stephanodiscus hantzschii* in the surface sample, comprising 50% of the assemblage. This taxon has a high TP optimum in the calibration set of 216 μg TP l^{-1} because it is found in the highest relative abundances in the lakes with the highest TP concentrations and the dataset includes many lakes with TP concentrations > 40 μg TP l^{-1}, thus resulting in an over-estimated value for Esthwaite Water. Furthermore, the surface sample (0-0.5 cm) could over-estimate the importance of *S. hantzschii* in the annual diatom assemblage as the half centimetre slice may represent only half a year (see chronology) and may simply represent a bloom in the species rather than an integrated annual assemblage.

It is interesting to note that the eutrophication process in Esthwaite Water is not reflected by the measured historical TP data. Data collected by Talling & Heaney (1988) shows that TP did not increase significantly over the period 1946-1986. The winter orthophosphate (and nitrate) data, however, revealed a clear picture of nutrient enrichment since the mid 1960s, with concentrations of c. 2 μg P l^{-1} in the 1940s and 1950s, rising to c. 5 μg P l^{-1} in the late 1960s, and rising more markedly to over 10 μg P l^{-1} since the mid 1970s. This implies some compensating shifts in the P fractions and led Talling & Heaney to believe that a strongly seasonal (mainly summer) internal loading had increased with productivity. A trend to more summer hypolimnetic anoxia and an increase in lake pH probably favoured P release from the sediments. Further evidence of recent nutrient enrichment was provided by the phytoplankton data, whereby *Stephanodiscus* spp. associated with eutrophic waters were prominently recorded in the basins of Esthwaite Water since 1970, although algal changes in response to enrichment were observed from the 1960s, probably due to overflowing septic tanks discharging into the lake (Talling & Heaney, 1988).

In summary, the palaeolimnological results are supported by the documented history of the lake, and the onset and rate of eutrophication as indicated by the diatom model corresponds closely to the timing of nutrient enrichment demonstrated by monitored biological and chemical data. Esthwaite Water has clearly experienced eutrophication over the last two decades following a long period of stable conditions, resulting in a change from a naturally mesotrophic lake to a meso-eutrophic one.

Figure 60 Aquatic macrophyte distribution map for Esthwaite Water



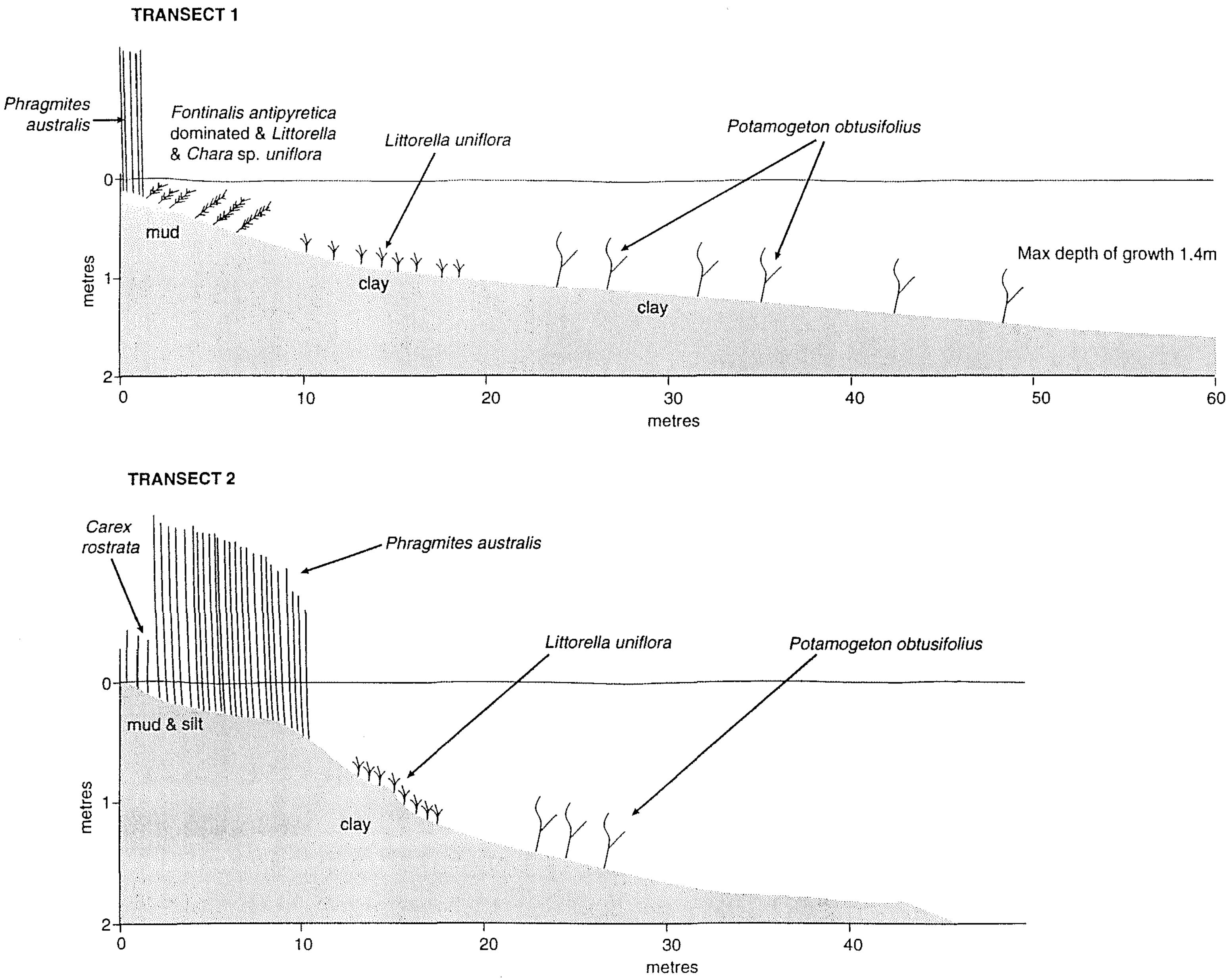


Figure 61 Aquatic macrophyte transect profiles for Esthwaite Water

Figure 62 Lithostratigraphic data for Esthwaite Water

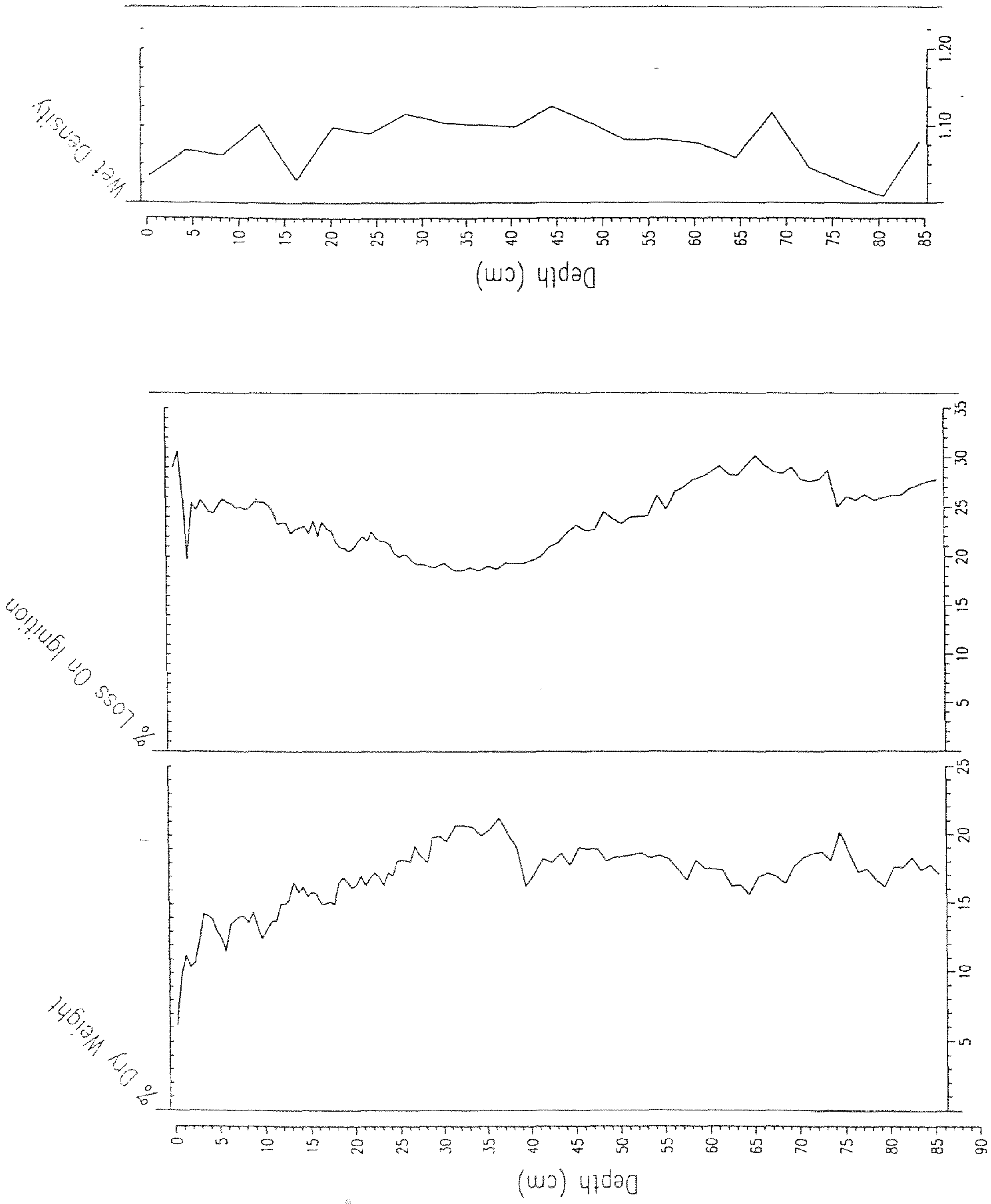


Figure 63 ^{210}Pb Activity versus Depth - Esthwaite Water

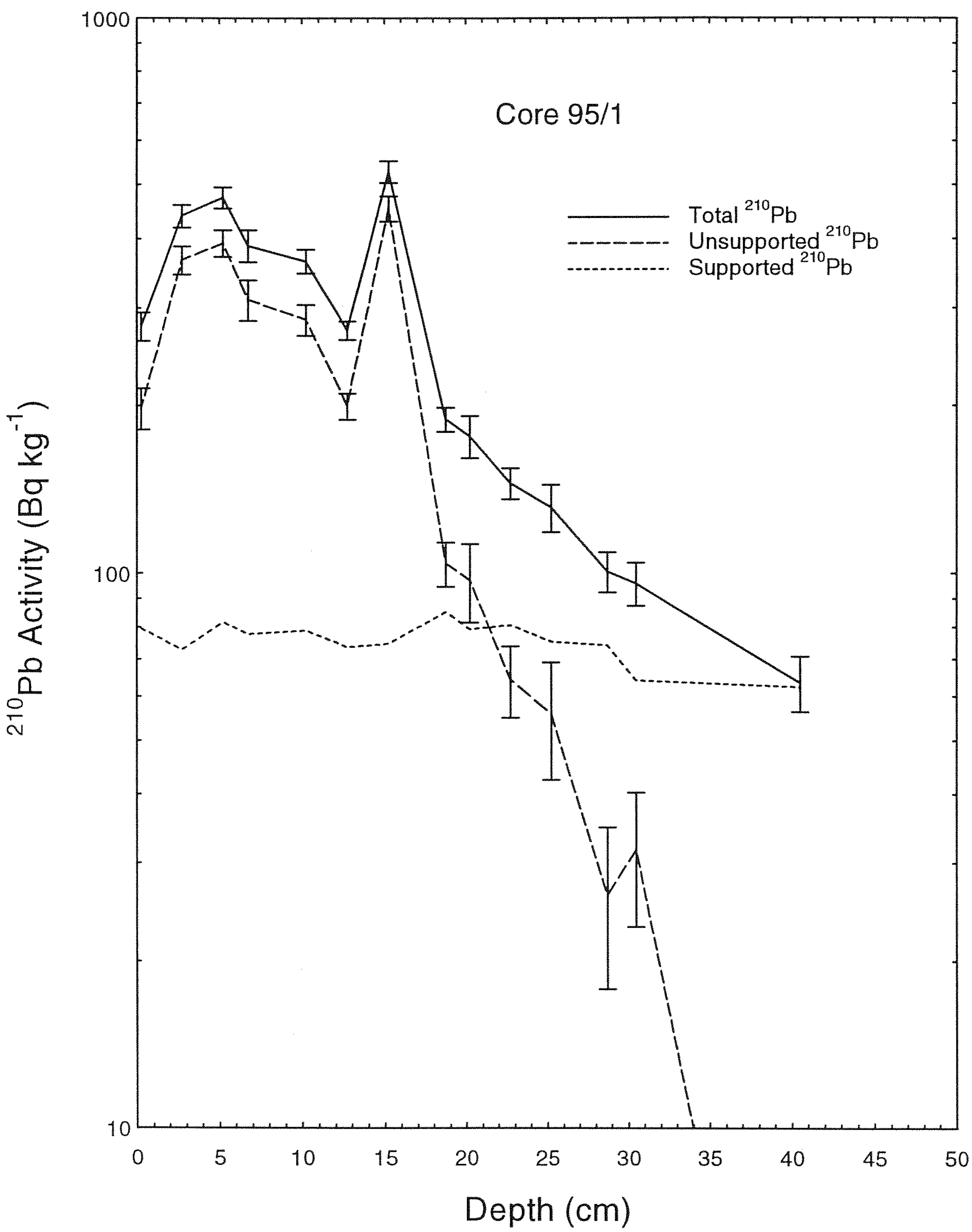


Figure 64 ^{137}Cs and ^{241}Am Activity versus Depth - Esthwaite Water

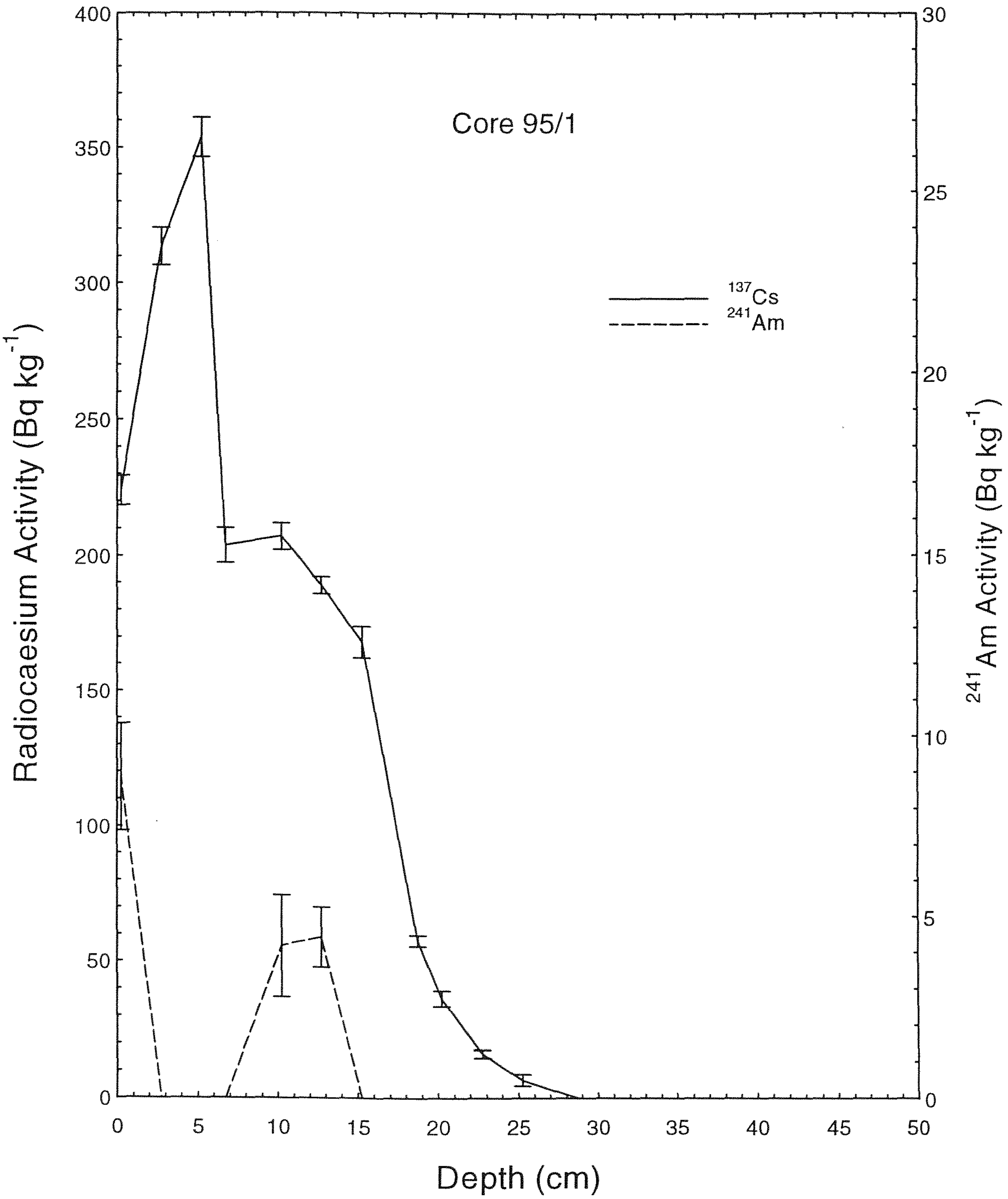


Figure 65 Depth versus Age - Esthwaite Water

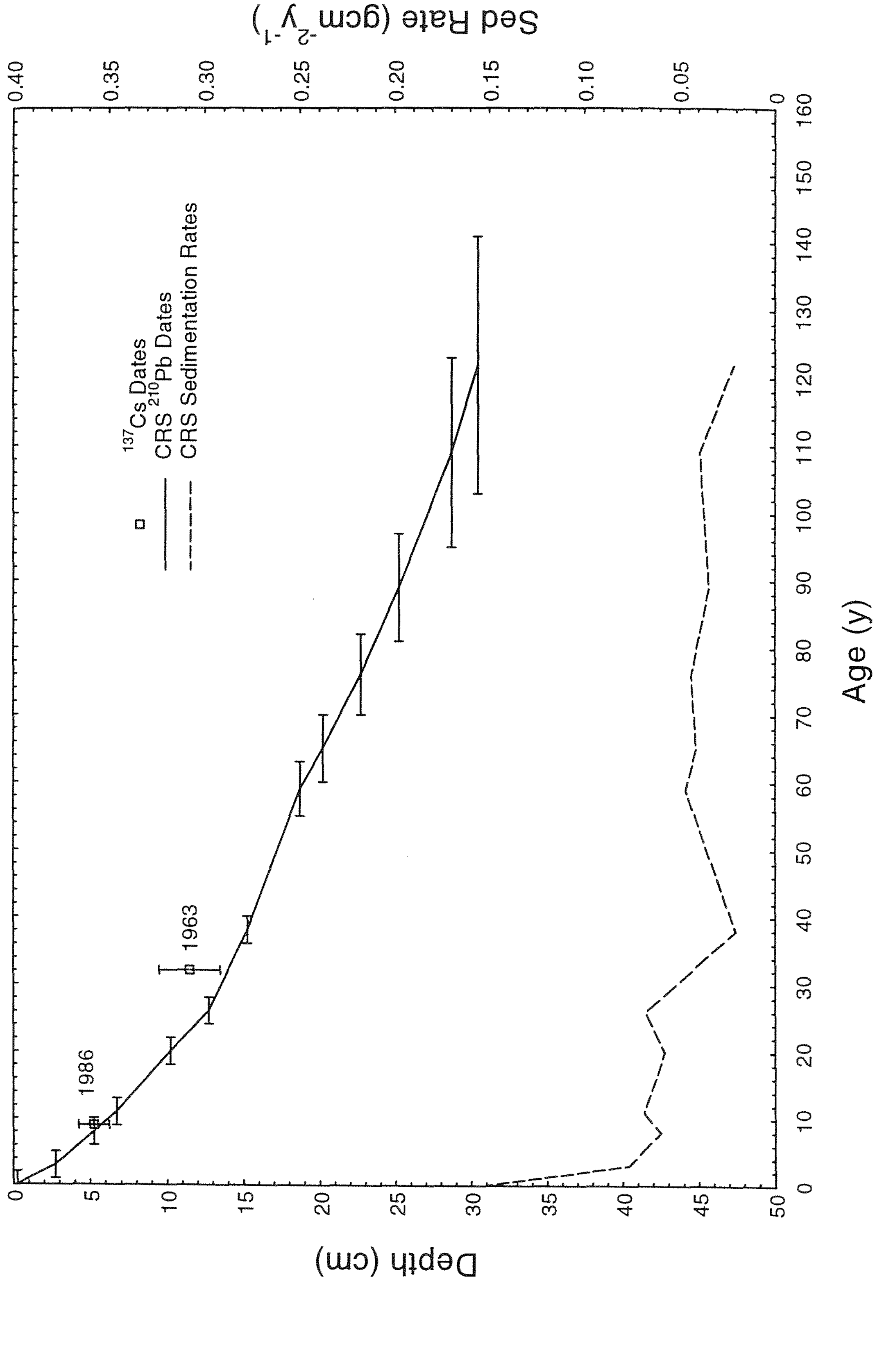
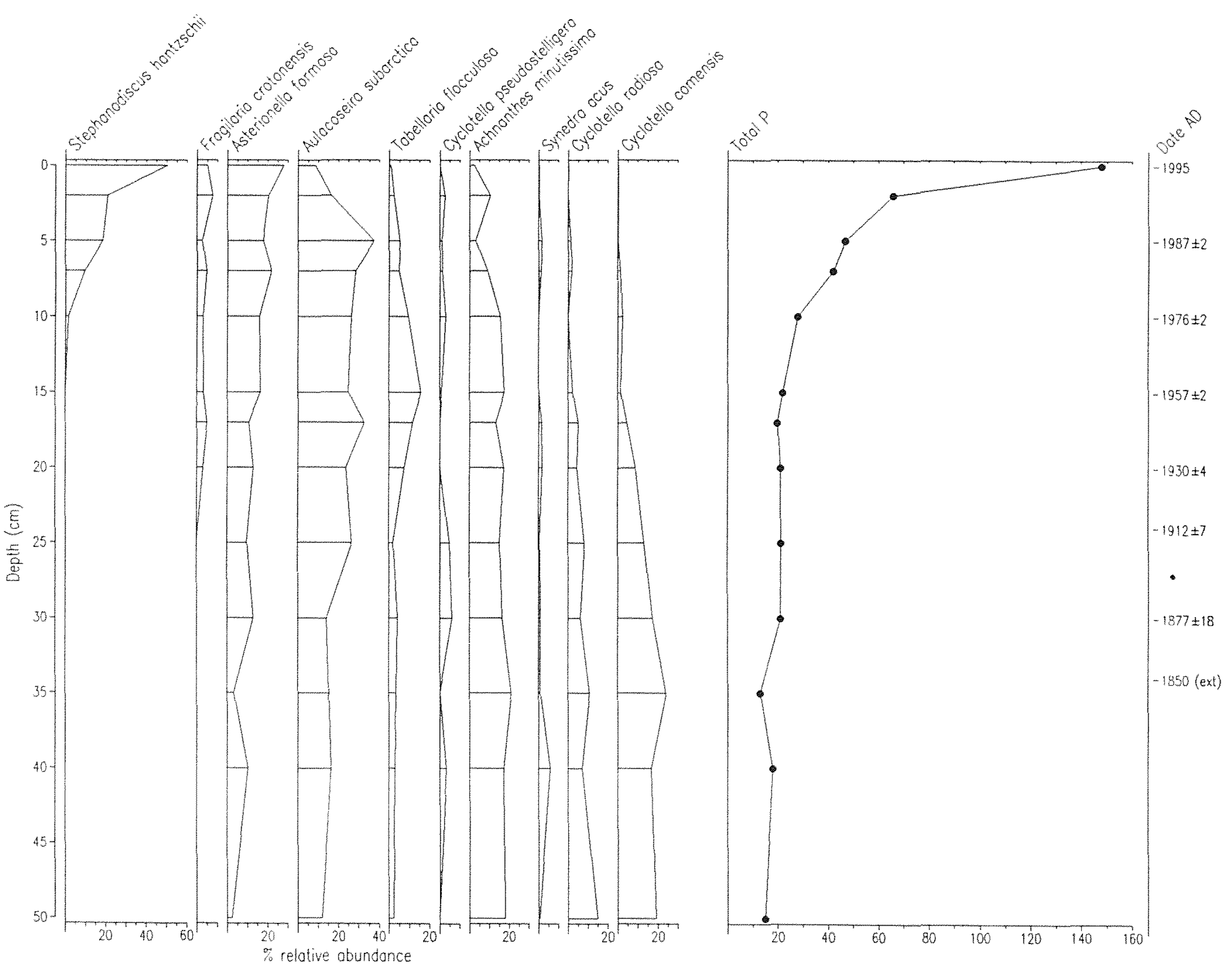
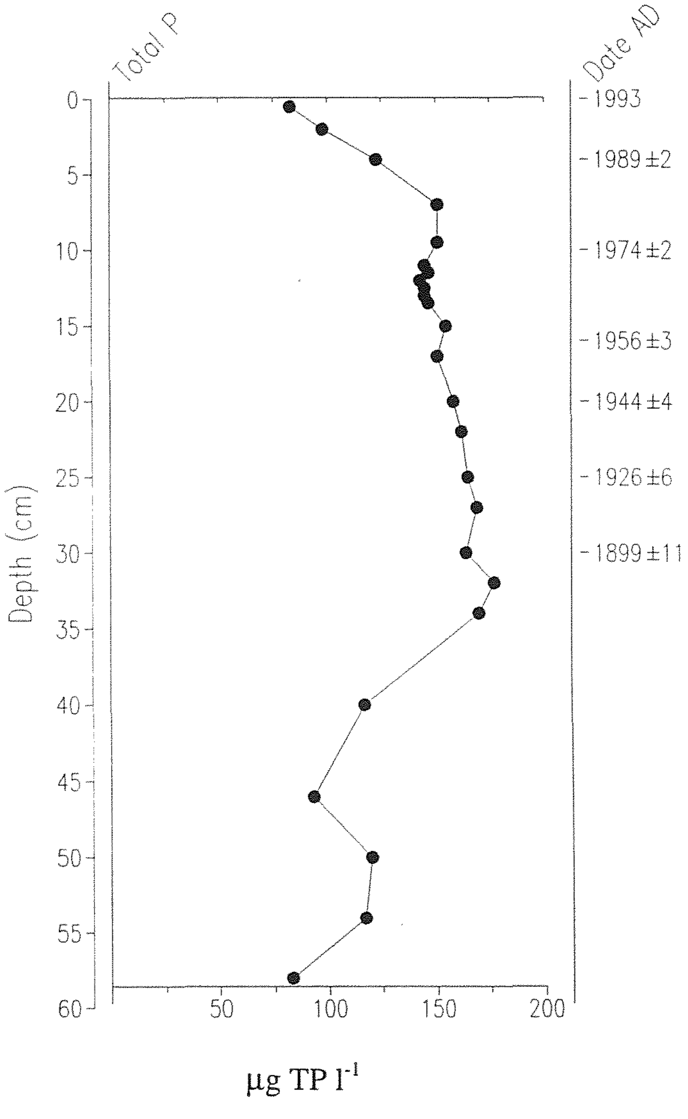


Figure 66 Summary diatom diagram and reconstruction for Esthwaite Water



Crose Mere

- The DI-TP results indicate that Crose Mere has always had high TP concentrations, classifying it as a naturally eutrophic lake.
- However, the lake has experienced an increase in TP concentrations since at least 1850 which is most likely associated with land-use changes in the catchment.
- The diatom model suggests that TP levels have been falling over the last 10 years but the exact reasons for this are not apparent.



μg TP l^{-1}

15. Crose Mere, Shropshire (SJ 430 305)

15.1 Site Description and Water Chemistry

- Altitude 92 m
- Area 15.2 ha (0.15 km^2)
- Maximum depth 9.5 m
- Formed in a kettlehole
- Geology is Keuper marl overlain by glacial sands and gravels
- Fed largely by groundwater
- The catchment is largely lowland agriculture with some fen pasture and deciduous woodland

pH	units	8.5
conductivity	$\mu S\ cm^{-1}$	414
alkalinity	mg $CaCO_3$	140
calcium	mg l^{-1}	66.3
total nitrogen	mg l^{-1}	1.4
total phosphorus	$\mu g\ l^{-1}$	111

15.2 Macrophyte Survey

Date of visit 3-7-96.
An aquatic macrophyte distribution map for Crose Mere is presented in Figure 67. The water was relatively clear at the time of visit, and a secchi disc depth of 4.3 m was recorded.

Submerged macrophytes are restricted to open areas of gradually sloping littoral with silty substrates at the east end of the Mere; the shoreline around the remaining perimeter is either steeply shelving, characterised by coarse substrates or dominated by reed beds. *Zannichellia palustris* is the most abundant submerged species, often growing in association with the alga *Hydrodictyon* sp., while another species of alga *Cladophora* sp. is most abundant in deeper water, growing to a water depth of approximately 2.2 m. *Potamogeton pectinatus* is occasional and locally frequent close to the outflow where it was most abundant at 0.3 m water depth. Small quantities of the charophyte *Chara* sp. were retrieved using the double-headed rake grapnel in open water in the north and south ends of the Mere.

The south side and two ends of Crose Mere are characterised by several stands of emergent swamp vegetation dominated by *Phragmites australis* and *Typha angustifolia* occasionally interspersed with small stands of *Scirpus lacustris* ssp. *lacustris* and *Cladium mariscus*.

The current flora of Crose Mere is indicative of a highly eutrophic system. The current dominance of the submerged flora by *Zannichellia palustris* is consistent with observations of this species at other nutrient rich lakes, e.g., Llangorse Lake in South Wales, where it briefly became dominant at a time of very low water transparency during a period of eutrophication (Monteith, 1996). *Z. palustris* appears to be well adapted to survive at low light levels (Bennion *et al.*, 1997) and it is possible that the degree of water transparency observed during the current survey is atypical of the recent history of this site.

It is quite clear from the documentary records that the aquatic macrophyte flora has undergone large changes in recent years (possibly most rapidly during the 1970s) and these are consistent with a process of eutrophication. Ratcliffe (1977), remarked on the presence of *Nuphar lutea*, *Nymphaea alba* and *Potamogeton crispus* at Crose Mere, none of which were found during the current survey. Generally however, the aquatic macrophyte species composition and distribution at Crose Mere appears to have changed little since a NCC survey conducted in September 1979. The main exceptions being the apparent impoverishment of an area of fen at the west end of the site which had contained emergent species such as *Sparganium erectum*. *Polyganum hydropiper* and *Iris pseudacorus*, none of which were found during this survey, and the absence of *Nuphar lutea*. A later NCC survey conducted in August 1987 found no submerged species, with the exception of a small amount of *Chara* sp.. It is possible that submerged plant growth at this time was almost entirely inhibited by water turbidity. A comparison of the surveys of 1979, 1987 and 1996 possibly reflect deterioration (in the 1980s) followed by slight improvement (in the 1990s) in transparency and thus water quality. However the extent of annual stochasticity in aquatic macrophyte assemblages in this type of system is poorly understood and any explanation of differences between surveys must be treated with caution.

Application of the species list presented in Table 18 classifies Crose Mere as type 10B (Palmer, 1992), a eutrophic category.

Table 18 Species list and DAFOR abundance rating for Crose Mere

	abundance	notes
Submerged taxa		
Potamogeton pectinatus	O	to 0.3 m water depth close to outflow
Zannichellia palustris	F	locally dominant in shallow water at west end
Chara sp.	R	in west and east ends
Hydrodictyon sp.	O	often in association with *Z. palustris*
Cladophora sp.	O	west end to 2.0 m water depth
Emergent taxa		
Cladium mariscus	O	within main emergent stands
Mentha aquatica	O	
Phragmites australis	F	several stands mainly on south side
Scirpus lacustris ssp. *lacustris*	O	within main emergent stands
Typha angustifolia	F	usually flanking *P. australis* on open water side
Alnus glutinosa		

15.3 Lithostratigraphy

A 120 cm sediment core (SCM03B) was taken from the deepest part of the lake in a water depth of 9.3 m using a piston corer on 22-6-93. The lower sediments (> 40 cm) were a very dark brown (10YR 2/2) detritus mud with some clay and silt (Ld3 Dg1 As+ Ag+ Lso+). Between 10-40 cm the sediment was a dark yellowish brown (10YR 3/4) and appeared to have a higher silt/clay content than the lower core section with some calcareous remains (Ld3 Ag1 As+ Lso+ Lc+ Dg+). The upper 10 cms were dark brown (10YR 3/3) flocculant, organic lake mud (Ld3 Lso1 Ag+).

The %dw and %loi profiles (Figure 68) show that the lower part of the core had a high organic content with %dw c. 15% and %loi c. 50%. The %loi values decreased to c.35% between 55-105 cm and decreased still further to c.20% between 10-40 cm, increasing again slightly in the upper 10 cms to c.30%. The %dw values increased slightly between 10-40 cm to c.20%. Similarly, the wd values were greater between 10-40 cm, indicating a period of denser, less organic sediments in this section of the core.

15.4 Radiometric Dating

^{210}Pb dates were calculated using both the CRS and CIC dating models (Appleby & Oldfield 1978). Above 16cm (dated ca.1956) there is little significant difference between the two models, both indicating a more or less constant sedimentation rate of 0.062±0.005 g $cm^{-2}y^{-1}$. Dates below 16cm are a little more problematical in view of the non-monotonic feature at c.20cm depth in the ^{210}Pb profile. Both models suggest that this feature records an episode of accelerated sedimentation in the mid 1940s, though they differ as to its duration and nature. The CRS model indicates a more prolonged event, with peak sedimentation occurring in ca.1943. The CIC model suggests a brief but more intense event such as a sediment slump. The dilution in ^{210}Pb activity is an argument in favour of the CRS model, though this is not conclusive. Both models indicate that prior to this event sedimentation rates were significantly lower than those determined for the post-1950 period. In the absence of any validating evidence, dates in the deeper sections of the core have been calculated using the average sedimentation rate of 0.039±0.009 g $cm^{-2}y^{-1}$ determined from both models. The results of these calculations are shown in Table 19 and the results are illustrated in Figure 69, Figure 70 and Figure 71.

The ^{137}Cs activity versus depth profile has a well defined peak at 14.25±2 cm that would appear to record maximum fallout from the atmospheric testing of nuclear weapons in 1963. This inference supported by the presence of traces of ^{241}Am at the same level (Appleby *et al.* 1991). The ^{137}Cs date is in excellent agreement with the ^{210}Pb chronology, which puts 1963 at a depth of 14 cm.

Table 19 Chronology of Crose Mere

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0.0	0.00	1993	0				
2.0	0.11	1991	2	2	0.063	1.04	9.7
4.0	0.25	1989	4	2	0.060	0.57	8.9
6.0	0.51	1985	8	2	0.060	0.45	9.3
8.0	0.79	1980	13	2	0.059	0.38	9.9
10.0	1.13	1974	19	2	0.061	0.36	10.7
12.0	1.49	1969	24	2	0.063	0.35	11.6
14.0	1.85	1962	31	3	0.055	0.31	14.9
16.0	2.21	1956	37	3	0.055	0.30	16.3
18.0	2.57	1950	43	4	0.063	0.35	21.0
20.0	2.94	1944	49	4	0.072	0.40	26.3
22.0	3.29	1938	55	5	0.058	0.32	26.5
24.0	3.63	1931	62	6	0.041	0.22	26.0
25.0	3.83	1926	67	6	0.039	0.19	
26.0	4.03	1921	72	7	0.039	0.19	
28.0	4.44	1910	83	9	0.039	0.19	
30.0	4.88	1899	94	11	0.039	0.19	

15.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in 25 levels of the upper 60 cms of the sediment core were calculated and Figure 72 illustrates the results for the major taxa. Diatom preservation was progressively worse downcore but did not prevent counting above 60 cm. A total of 80 taxa was observed, 66 of which were present in the calibration set. All of the common taxa were well represented in the calibration set, with greater than 97% of the fossil assemblage being used in the calibration procedure for all except the bottom sample (91%).

Figure 72 illustrates that there have been marked changes in the diatom species composition over the period represented by the 60 cm core (post-1750). The lower samples (pre-1850) were all dominated by mesotrophic to eutrophic, planktonic taxa, particularly *Cyclotella ocellata, Cyclotella radiosa, Aulacoseira granulata, Stephanodiscus parvus* and two non-planktonic *Fragilaria* spp. (Note that this is a very similar assemblage to that in the lower samples of the Betton Pool core). There was an expansion of *Stephanodiscus parvus* and consequent decline in *Cyclotella ocellata* during the late nineteenth century. *Stephanodiscus parvus* was the dominant species throughout the early and mid 1900s, and a number of other eutrophic taxa increased in importance in the early 1900s, including *Aulacoseira granulata, Aulacoseira ambigua, Stephanodiscus hantzschii* and *Stephanodiscus neoastraea*. The assemblage changed again slightly in the period 1940-1970 when *Fragilaria crotonensis* and *Asterionella formosa* were present in higher relative abundances than in previous years, taxa with lower TP optima than the *Stephanodiscus* spp. The assemblage has changed significantly since c.1980. *Fragilaria* *crotonensis* and *Asterionella formosa* have continued to expand and have replaced *Stephanodiscus parvus* as the dominant species.

15.6 Total Phosphorus Reconstruction

The TP reconstruction indicates that the lake has been eutrophic in terms of TP concentrations since at least 1750 with concentrations always in excess of 80 μg TP l^{-1}. The model infers that TP concentrations increased in the late nineteenth century and remained high throughout most of the 1900s. However, the model suggests that TP levels began to decline again c.1982 and have continued to do so to the present day. For example, the diatom-inferred TP concentrations were 83 μg TP l^{-1} in c.1750 (58 cm), 117 μg TP l^{-1} in c.1850 (40 cm), increasing to 170 μg TP l^{-1} by c.1880 (35 cm), stabilised at c. 140-170 μg TP l^{-1} during the twentieth century (1900 and 1980), decreased to 123 μg TP l^{-1} by 1989 and declined still further to 83 μg TP l^{-1} by 1993. Thus, after a period of marked enrichment, TP levels have returned to pre-enrichment concentrations.

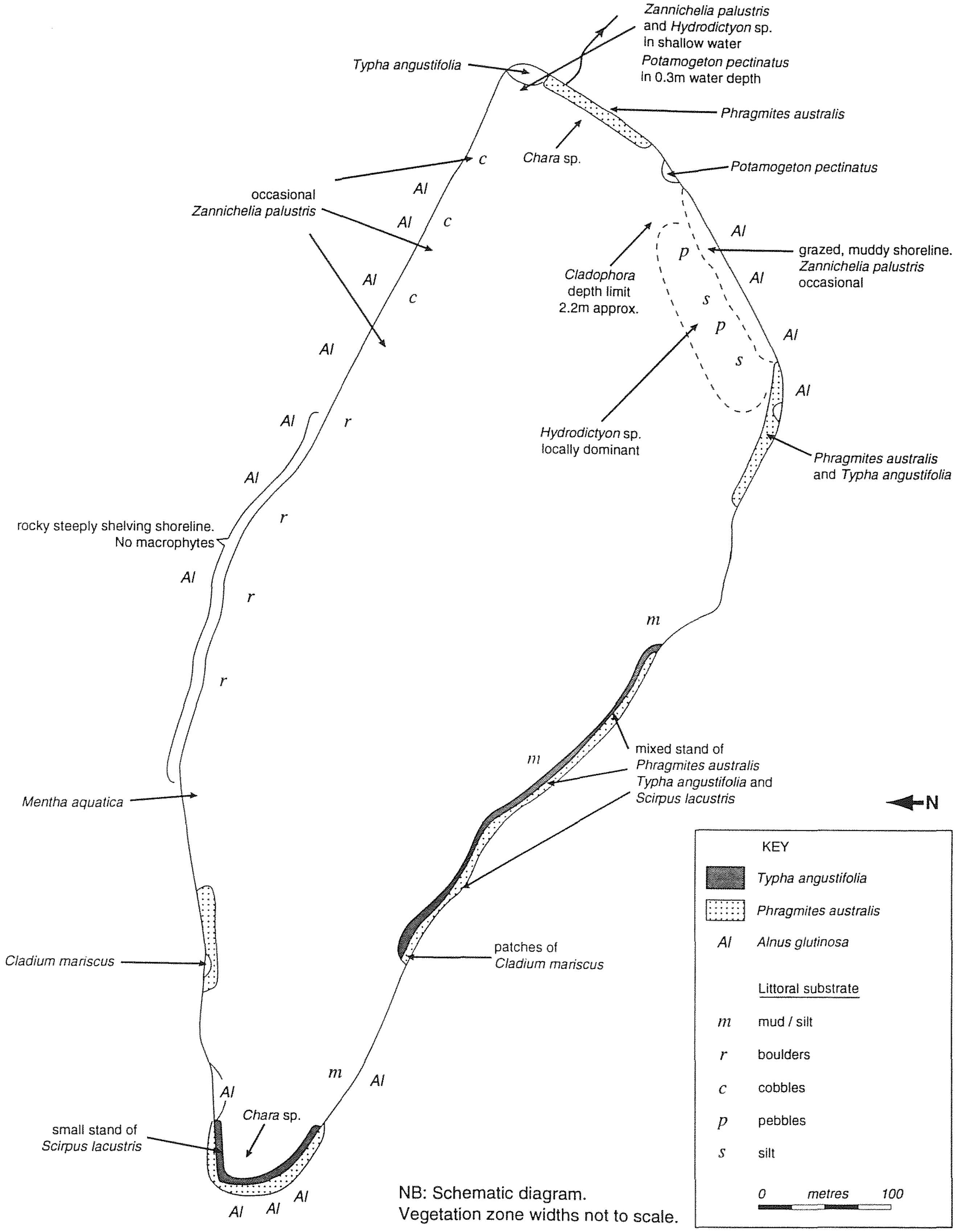
15.7 Discussion

The post-1850 eutrophication at Crose Mere, as inferred from the diatom model, is most likely related to changes in land-use in the lake catchment, although there are no documented data to support these conclusions. The lake is currently surrounded by open rough pasture used for cattle grazing. Parish data from 1931 indicate that the catchment saw an increase in cattle, pig and sheep numbers and an increase in arable land between 1931 and 1987 (Moss *et al.*, 1992), but there are no earlier available data. The introduction of nitrogen-based fertilisers, increase in the production of slurry and an increase in the Canada geese population have also been suggested as possible nutrient sources to the lake. Therefore the increase may be attributed to agricultural expansion in the catchment. There are also no available data documenting any land use or management changes that might explain the reduction in TP concentrations since c.1982 as inferred by the diatom model. Moss *et al.* (1992), however, report the increasing importance of zooplankton grazing in the lake linked to lower fish stocks, which could reduce the crops of easily digested, small, planktonic diatoms, such as *Stephanodiscus* spp., and would favour the survival of the less digestible, long diatom forms such as *Fragilaria crotonensis* and *Asterionella formosa,* which dominate the assemblages today.

The diatom-inferred TP concentration of 83 μg TP l^{-1} for the surface sample does in fact appear to slightly under-estimate the annual mean TP concentrations of the lake, although it falls within the seasonal range. Water chemistry data collected by the EA during 1995/6, gave an annual mean TP value of 111 μg TP l^{-1} and a range of 10-204 μg TP l^{-1} over the sampling period. The model under-estimates because of the importance of *Fragilaria crotonensis* and *Asterionella formosa* in the surface sample, taxa which are generally only observed at such high frequencies in lakes with lower TP concentrations than Crose Mere. This may be because of the grazing impacts (see above) or perhaps linked to the importance of nitrogen limitation in the mere's rather than phosphorus, which is generally the limiting nutrient in temperate freshwater but which is in plentiful supply in the lakes of this region. As for Betton Pool, Crose Mere would be classed as a eutrophic lake based on its high TP concentrations, although such values are typical for the Cheshire and Shropshire meres.

In summary, the diatom model indicates that Crose Mere has always been a eutrophic lake but has experienced an increase in TP concentrations since 1850, most likely due to human activity in the catchment. The model suggests that TP concentrations have been falling over the last ten years but the changes in the diatom assemblages may be related more to changes in grazing pressure or even other nutrients than to any real change in lake TP levels. Moss *et al.* (1992) concluded that Crose Mere did not appear to be at any greater threat from the impacts of eutrophication than one might expect for an agricultural region. Continued monitoring of nutrient concentrations in the lake would be desirable based on the palaeolimnological findings of the current study.

Figure 67 Aquatic macrophyte distribution map for Crose Mere



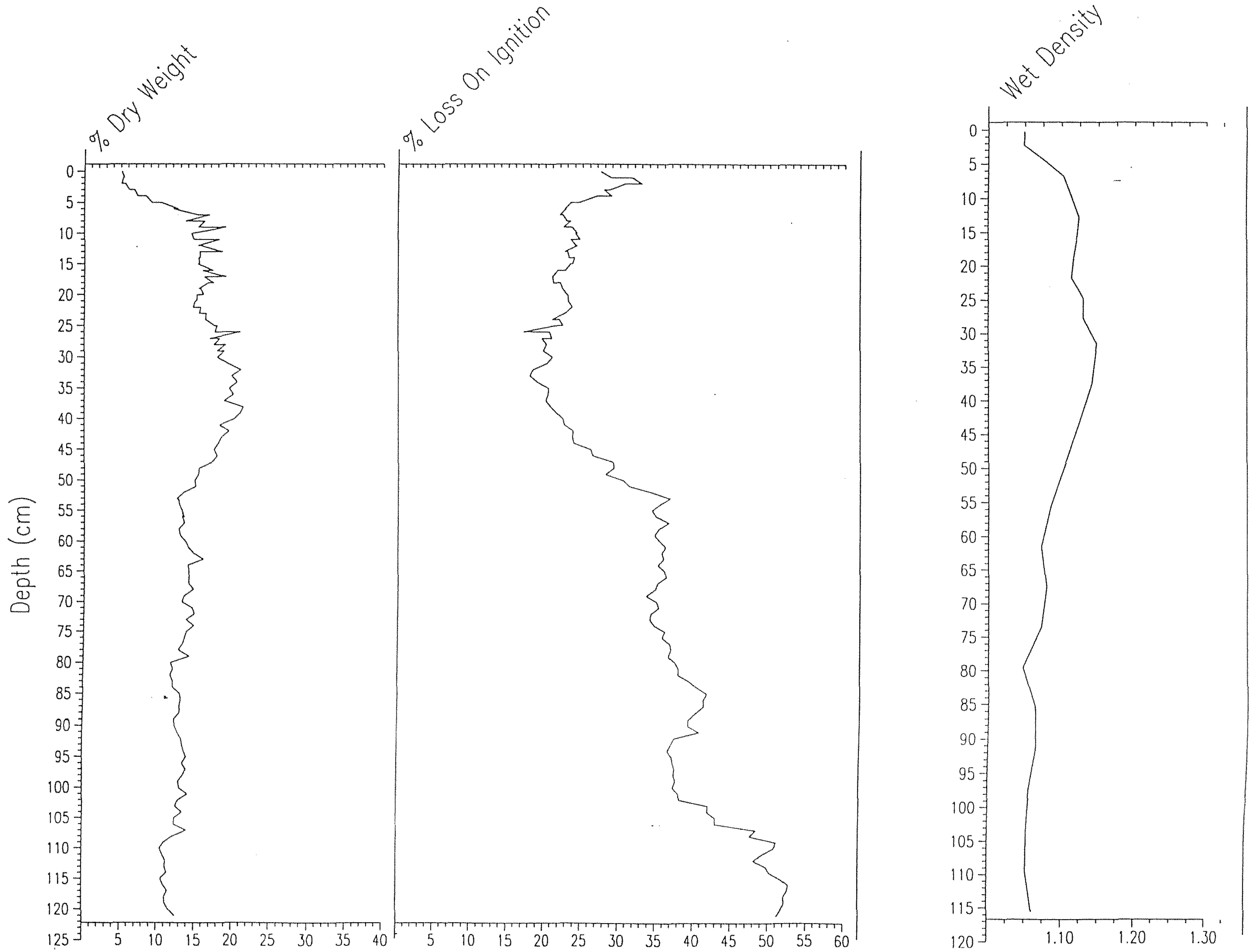


Figure 68 Lithostratigraphic data for Crose Mere

Figure 69 ^{210}Pb Activity versus Depth - Crose Mere

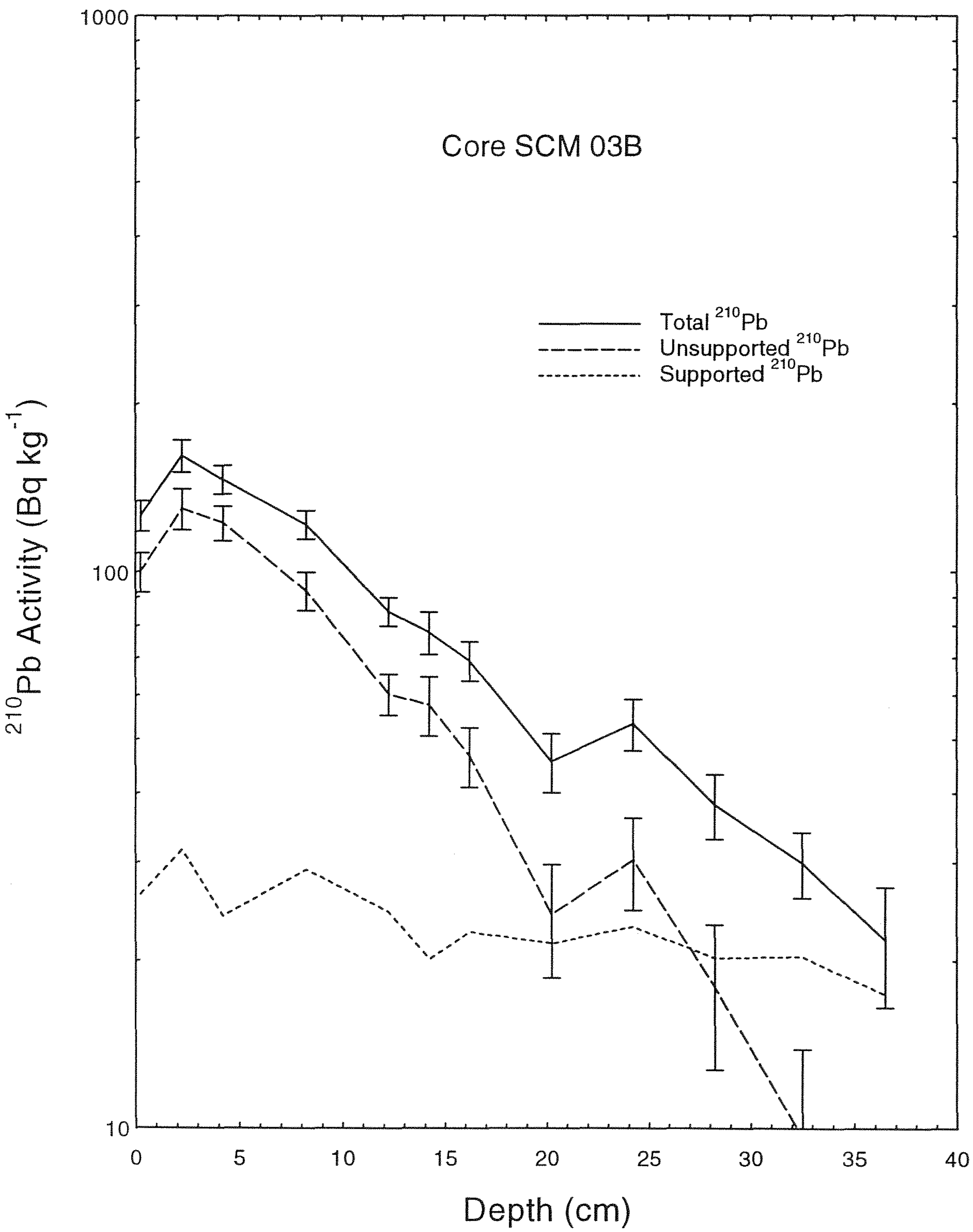


Figure 70 ^{137}Cs and ^{241}Am Activity versus Depth - Crose Mere

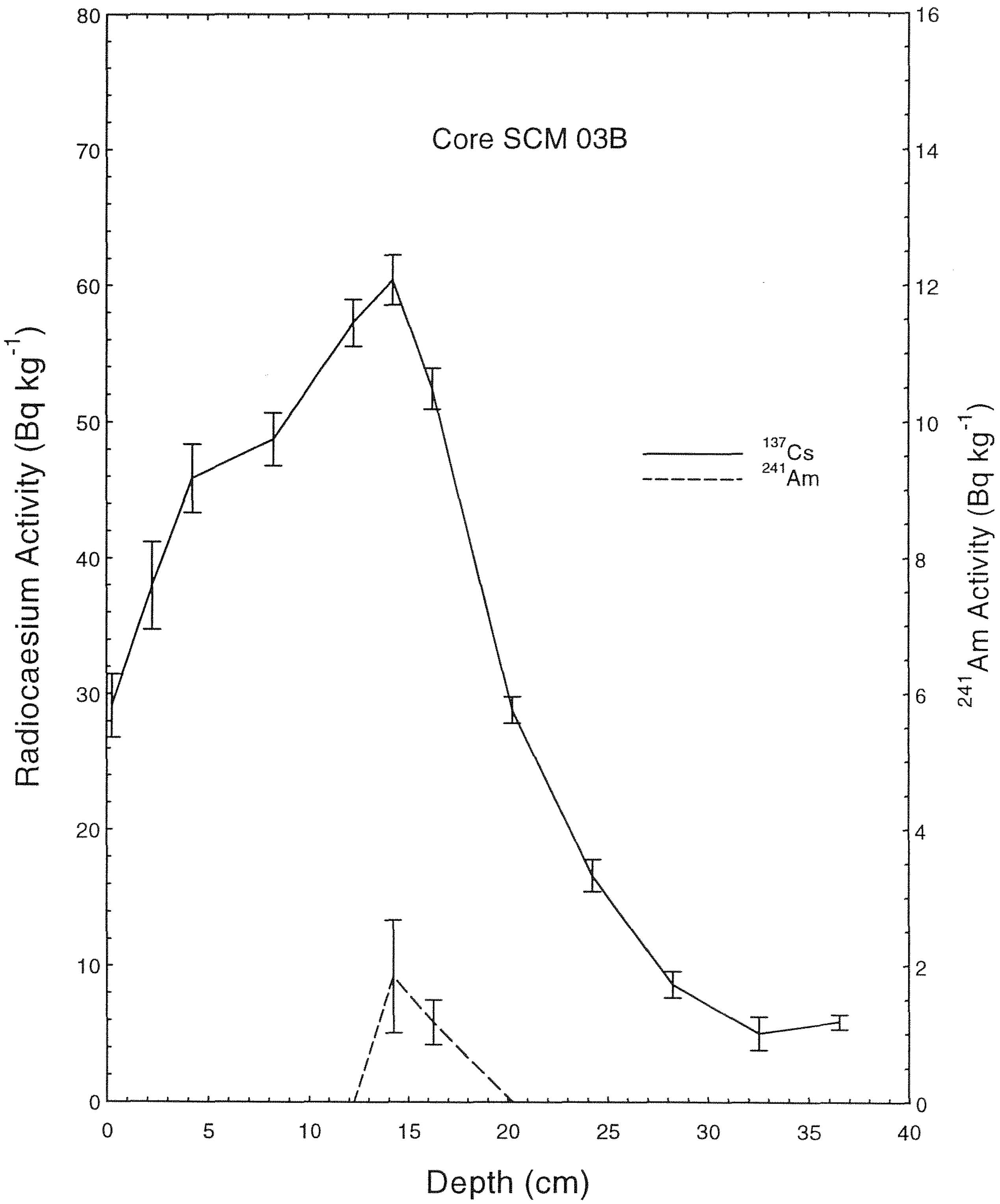


Figure 71 Depth versus Age - Crose Mere

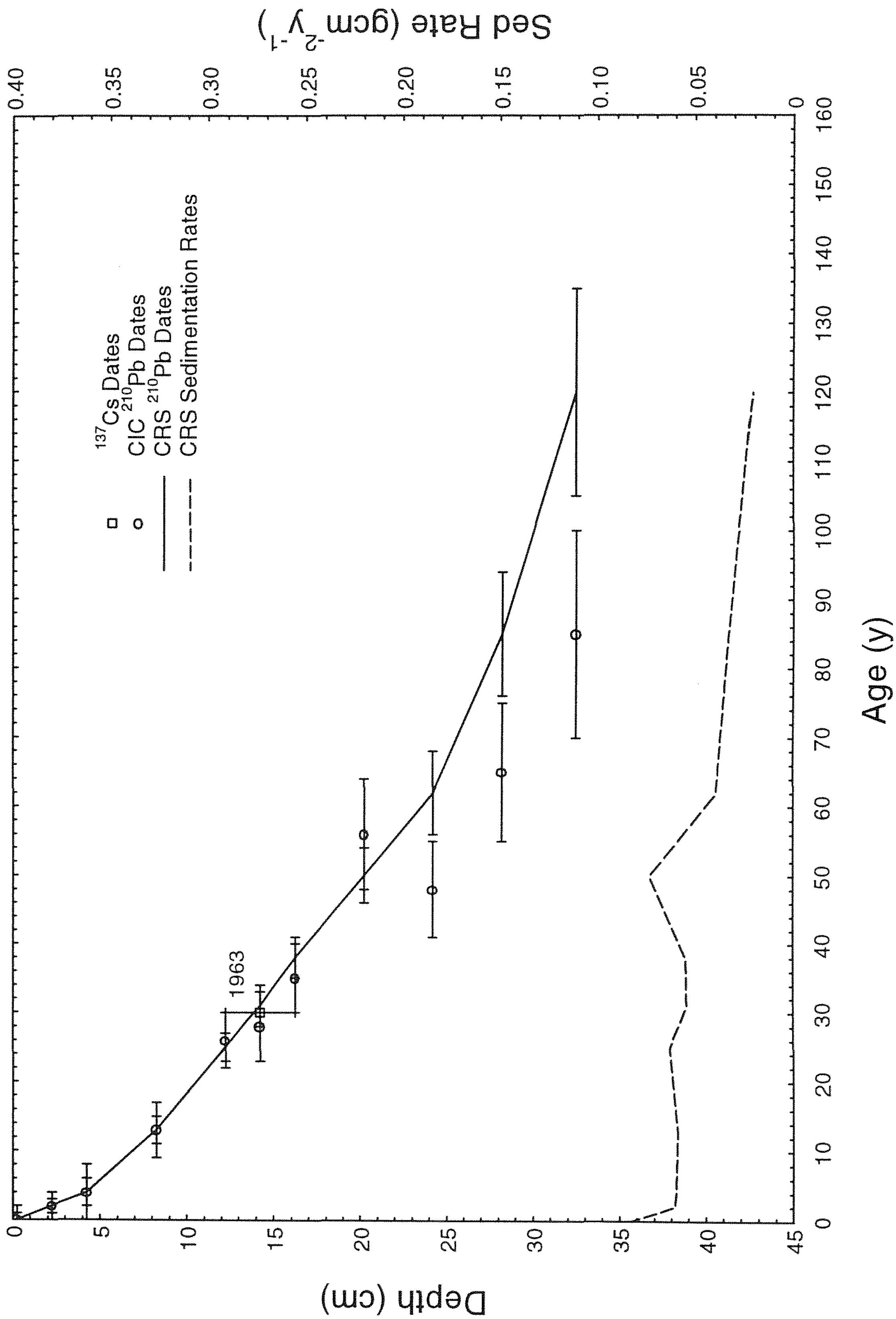
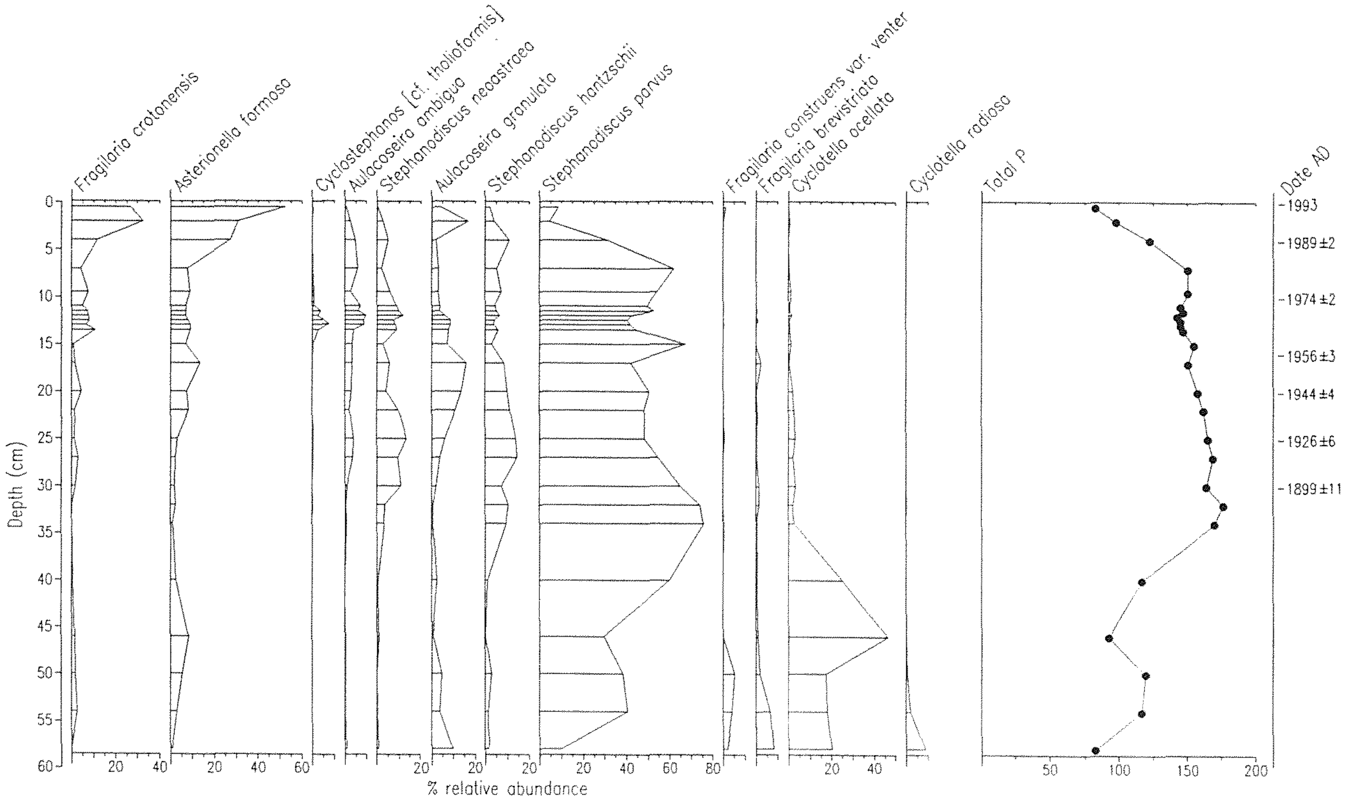
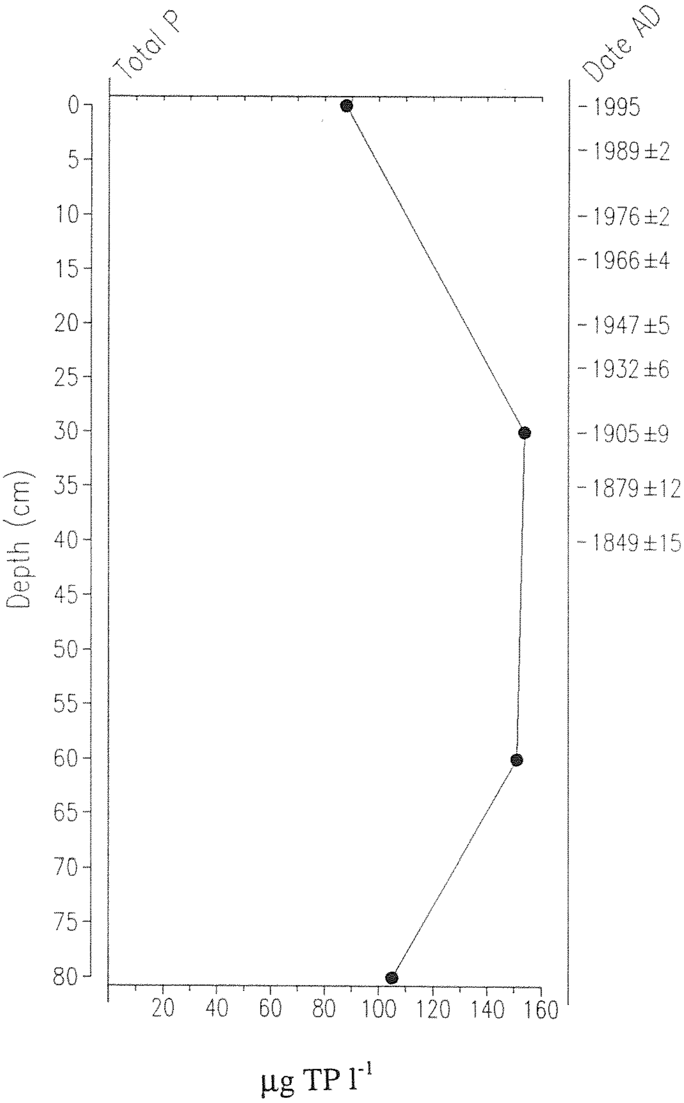


Figure 72 Summary diatom diagram and reconstruction for Crose Mere



Malham Tarn

- **The DI-TP results suggest that Malham Tarn is a naturally eutrophic lake and has not experienced any major increase in TP concentrations since 1800.**
- **However, these data must be interpreted with caution in view of the unique chemistry of the tarn's water and sediments.**



16. Malham Tarn, North Yorkshire (SD 894 668)

16.1 Site Description and Water Chemistry

- Malham Tarn is situated in a predominantly limestone area of the Pennine uplands.
- Altitude 375 m
- Area 61 ha
- Mean depth 2.4 m
- Maximum depth 4.4 m
- Catchment area 11.5 km2
- Land use is predominantly unimproved grassland with some improved pasture

pH	units	8.3
conductivity	$\mu S\ cm^{-1}$	218
alkalinity	mg $CaCO_3$	103
calcium	$mg\ l^{-1}$	42.3
total nitrogen	$mg\ l^{-1}$	0.9
total phosphorus	$\mu g\ l^{-1}$	21

nb. These readings were taken from the River Aire outflow.

16.2 Macrophyte Survey

Date of visit 10-6-95.
An aquatic macrophyte distribution map and transect profiles for Malham Tarn are provided in Figure 73 and Figure 74. Water clarity was very good at the time of the survey and the secchi disc could be seen at the maximum depth of the lake.

The north and east shorelines are characterised by a boulder/cobble substrate with *Fontinalis antipyretica* attached to cobbles above the water-line and occasional shoreline *Caltha palustris* and *Mentha aquatica*. The littoral appears to be too exposed to allow the establishment of macrophytes. However, *Elodea canadensis* thrives at the stone weir outflow. A stand of low growing *Chara* sp. is present in shallow water off the south-west shore, on a sand/silt substrate to a depth of approximately 0.6m.

The north-west shore is characterised by an overhanging peat margin, and an old "pre-weir" peat terrace, now submerged by approximately 1m, supports little vegetation other than occasional small stands of *E. canadensis* in sheltered hollows. A detached strand of the acidophilous species *Callitriche hamulata* was found in the lake outflow and it is likely that this would have originated in this part of the lake.

The area around the lake outflow was dominated by *Salix* scrub, flanked by *Carex rostrata*, *Menyanthes trifoliata*, *Caltha palustris*, *Iris pseudacorus* and *Mentha aquatica*.

Numerous transects across the lake employing both a double-headed rake and bathyscope revealed alternating dense stands of *Chara* sp. and *E.canadensis* in similar proportions. There was no clear pattern in the local distribution of these two species, with the exception that *E.canadensis* dominated in deeper water off the peat shoreline and the zone around the weir. *E.canadensis* clearly thrives in the Malham Tarn environment; the surveyor has never encountered such substantial cover/biomass or individual plant size of this species before.

In the deepest part of the tarn, *E.canadensis* and *Chara* sp. are dominant to a depth of approximately 3.6 m, with the exception of an isolated stand of *Potamogeton lucens* which occurs to a maximum depth of 3.4 m. This was the only location in which *P.lucens* was found. Balls of *Cladophora* sp., up to 7 cm in diameter, were observed covering a significant area of the lake bed towards the outflow, at a water depth of approximately 2 m.

The aquatic macroflora of Malham Tarn has been well documented over recent years and it is clear that major changes have taken place in terms of distribution, relative abundance and presence/absence. In his description of the site, Holmes (1965) describes a diverse aquatic macroflora which includes the *Potamogeton* taxa, *P. perfoliatus* and *P. berchtoldii*, and *Myriophllum spicatum,* none of which were found during the current survey. In addition *Potamogeton lucens* was said to occur in several large patches, whereas it now only occurs in one isolated patch in the deepest water. Interestingly, *Elodea canadensis* only appeared in 1962, by a chance introduction, (perhaps from a collecting net of a visiting student). It is possible that vegetational change at Malham Tarn has largely been brought about largely as a result of the change in the competitive environment brought about by the introduction of this alien species which thrives in the Tarn today. The possible effect of nutrient enrichment at the site is therefore difficult to ascertain.

Applying the species list presented in Table 20, Malham Tarn is classified as Type 5A (Palmer, 1992), a mesotrophic category.

Table 20 Species list and DAFOR abundance rating for Malham Tarn

	abundance	notes
Submerged taxa		
Cladophora sp.	O	Locally abundant in shallow water
Chara sp.	A	Locally dominant throughout
Fontinalis antipyretica	F	Shoreline rocks in north and east
Elodea canadensis	A	Locally dominant throughout
Callitriche hamulata	R	Single detached specimen found
Emergent taxa		
Equisetum palustre	R	South-east shoreline
Potamogeton lucens	R	Single stand in deep water
Carex rostrata	O	Locally abundant
Menyanthes trifoliata	R	Near outflow
Iris pseudacorus	R	Near outflow

16.3 Lithostratigraphy

An 83 cm sediment core (MALH2) was taken from the deepest part of the lake at a water depth of 4 m using a Mackereth corer on 10-6-95. The bottom 10 cms of the core were a very dark grey (10YR 3/1) inorganic, marly mud with fine sand (Ld2 Lc1 Ag1 Ga+ Lso+ Dh++). Above this the sediment changed to a black (10YR 2/1), slightly more organic, silty lake mud with herbaceous plant fragments (Ld3 Ag1 Lso+ Lc+ Dh++). There was a sandy layer from 2-15 cm (Ld3 Ga1 Dh++ Lso+ Lc+) . The upper few cms were a very dark grey (10YR 3/1), more organic lake mud (Ld3 Lso1 Dg+ Dh+ Lc+ Ga+) with traces of marl and fine sand.

The %dw and %loi profiles (Figure 75) show that the sediments of Malham Tarn are largely inorganic, although there was a progressive increase in percentage organic matter from the core base to 15 cm, with %dw decreasing from 35% to 20% and %loi rising from 12% to 30%. The sandy layer was apparent at 2-15 cm with a marked decrease in %loi to 15%. The changes in wd were similar to those in %dw with a gradual decrease from 1.25 g cm^{-3} at the base to 1.05 g cm^{-3} at c. 20 cm, and a slight increase again between 2 and c.15 cm marking the increase in sand content.

16.4 Radiometric Dating

The Malham Tarn core had a relatively simple ^{210}Pb profile. Calculations using the CRS and CIC ^{210}Pb dating models both indicate a more or less uniform sedimentation rate prior to c.1970 of 0.046 ± 0.003 g cm^{-2}y^{-1}. There may however have been a significant increase during the past 30 years.

The ^{137}Cs profile had two peaks that appear to record fallout from the 1986 Chernobyl accident and 1963 weapons test fallout maximum. Within the limits of the resolution of the ^{137}Cs peaks these dates are in reasonable agreement with the ^{210}Pb results.

The results are summarised in Table 21 and in Figure 76, Figure 77 and Figure 78. The well resolved 1963 ^{137}Cs date has however been used to make a small correction to the ^{210}Pb dates.

Table 21 Chronology of Malham Tarn

Depth		Chronology			Sedimentation Rate		
		Date	Age				
cm	g cm^{-2}	AD	y	±	g cm^{-2} y^{-1}	cm y^{-1}	± (%)
0.0	0.00	1995	0				
2.0	0.26	1993	2	2	0.110	0.75	8.3
4.0	0.58	1989	6	2	0.091	0.55	8.7
6.0	0.94	1985	10	2	0.081	0.45	7.2
8.0	1.30	1981	14	2	0.083	0.44	9.2
10.0	1.69	1976	19	2	0.075	0.40	6.9
12.0	2.07	1971	24	3	0.078	0.40	13.9
14.0	2.45	1966	29	4	0.079	0.40	12.4
16.0	2.80	1961	34	5	0.069	0.38	13.7
18.0	3.14	1955	40	5	0.059	0.29	17.2
20.0	3.50	1947	48	5			
22.0	3.88	1939	56	6			
24.0	4.26	1931	64	6			
26.0	4.65	1922	73	7	0.046	0.24	
28.0	5.06	1914	81	8			
30.0	5.46	1905	90	9			
35.0	6.66	1879	116	12			
40.0	8.02	1849	146	15			

16.5 Diatom Stratigraphy

The percentage relative frequencies of diatom species in four levels of the sediment core were calculated and Figure 79 illustrates the results for the major taxa. Diatom preservation was generally poor, owing to the calcareous nature of the lake sediments. A total of 72 taxa was observed, 54 of which were present in the calibration set. All of the common taxa were well represented in the calibration set, with greater than 90% of the fossil assemblage being used in the calibration procedure.

Figure 79 illustrates that there have been slight changes in the diatom species composition in terms of relative percentages over the period represented by the 80cm core (c.1700-1995). However, there has been no clear species replacement and the assemblages have been dominated by the same major, non-planktonic taxa throughout the core: *Achnanthes minutissima*, *Amphora pediculus*, and the benthic *Fragilaria* spp., all species commonly found attached to either the sediments, stones or macrophyte surfaces of shallow, alkaline waters. The only changes were that *Fragilaria construens* var. *binodis* and *Fragilaria lapponica* accounted for greater relative abundances in the 30 cm sample (c. 1900) than in the upper and lower samples, *Gyrosigma acuminatum* occurred in the highest frequencies in the bottom sample (c. 1700), and *Achnanthes minutissima* had higher frequencies in the bottom and surface sample than in either the 60 cm (c. 1800) or the 30 cm sample (c. 1900).

16.6 Total Phosphorus Reconstruction

The TP reconstruction shows that the lake has had high TP concentrations throughout the period represented by the sediment core (1700-1995), which would place the lake in the eutrophic category. However there was no clear trend in the reconstructed history. The model provided an estimate of 105 μg TP l^{-1} for c. 1700 (80 cm), then indicated a slight increase to c. 150 μg TP l^{-1} for c. 1800 (60 cm) and c.1900 (30 cm), followed by a decrease to 88 μg TP l^{-1} by 1995.

16.7 Discussion

The results suggest that Malham Tarn is a naturally eutrophic lake and has not experienced any major increase in TP concentrations since 1800. However, these data must be viewed with caution in view of the unique chemistry of the tarn's water and sediments. It is a highly alkaline site, rich in calcium carbonate with frequent silica limitation, and pH is rarely below 8.0. Such conditions can only be tolerated by a small number of species thus restricting the flora, and may have more influence on the diatom assemblages than open water TP concentrations.

The lack of planktonic forms in the Malham Tarn sediments causes uncertainty over the results of the model because the relationship between non-planktonic taxa and open water nutrient concentrations is not as direct as that for planktonic forms (Bennion, 1995). The absence of such forms, particularly the centric diatoms, may be related to the need for motility on the sediments because these are non-motile taxa and would be buried in the deposits whenever there was disturbance by wave action etc and would be unlikely to survive (Round, 1953). The development of epiphytic (attached to plant substrates) and epilithic (attached to stones) communities at the expense of planktonic ones may be regarded as one of the characteristics of a calcareous, eutrophic system (Round, 1953). Furthermore, Malham Tarn's calcareous sediments are naturally inorganic with traces of sand and a lack of organic remains. The importance of *Amphora pediculus* in the fossil assemblages is most likely explained by the nature of the sediment because it is a species commonly observed attached to sand grains.

These factors may explain why the model over-estimates TP values for Malham Tarn. In the EA 1995/6 survey, TP concentrations in the outflow ranged from 9-32 μg TP l^{-1} with a data set mean of 21 μg TP l^{-1}, whereas the model estimates a current TP value for the lake of 88 μg TP l^{-1}. An earlier study of Malham Tarn discussed the difficulty with classifying the lake within the usual freshwater trophic system because it exhibits eutrophic features, such as high nutrient status, high submerged macrophyte production and shallow water depth, and also oligotrophic features, such as low phytoplankton production and the absence of marginal reed beds (Round, 1953). This study concluded that the chemical and physical nature of the sediment, combined with the chemical composition of the water, was the governing factor in the composition of the algal floras.

Figure 73 Aquatic macrophyte distribution map for Malham Tarn

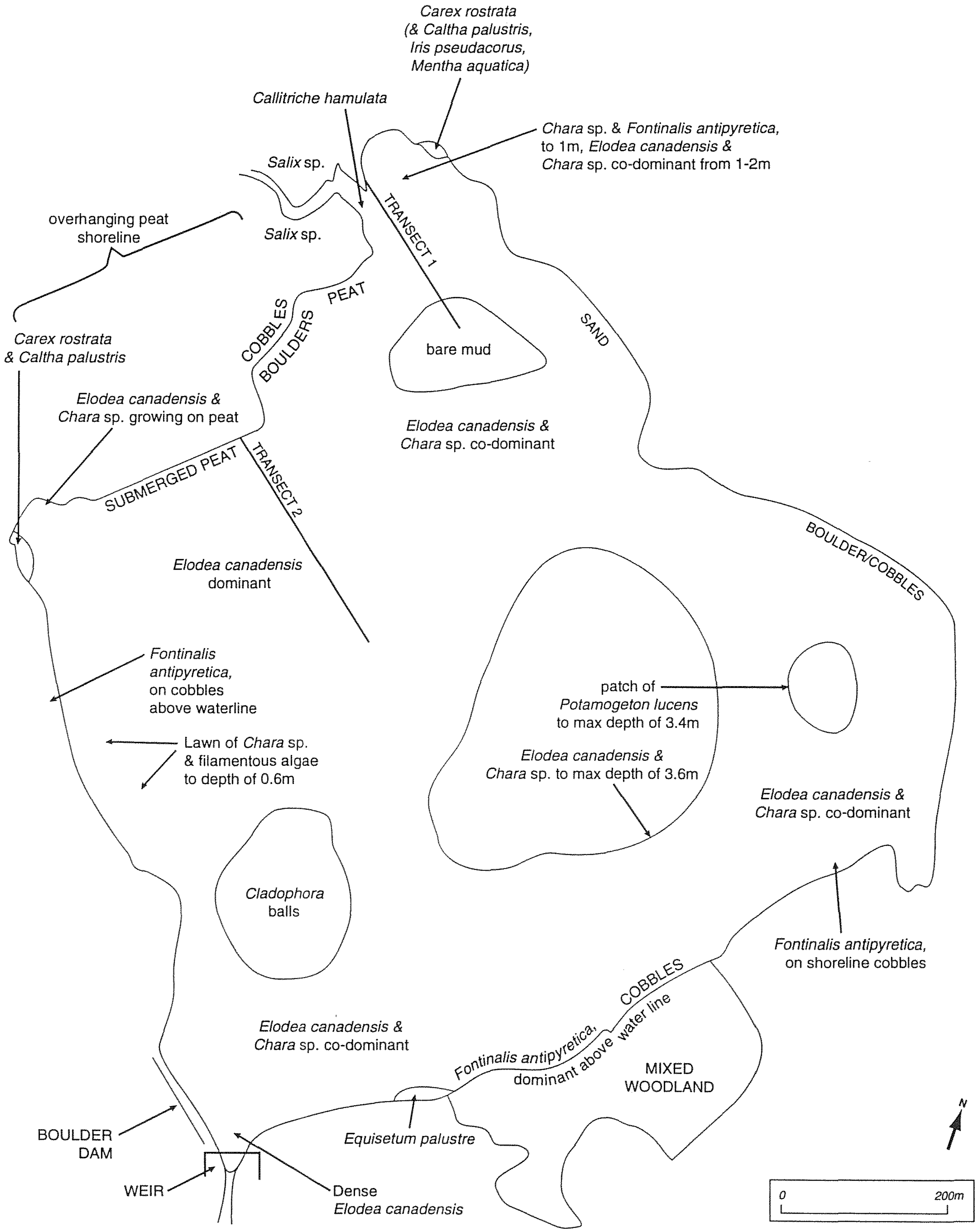
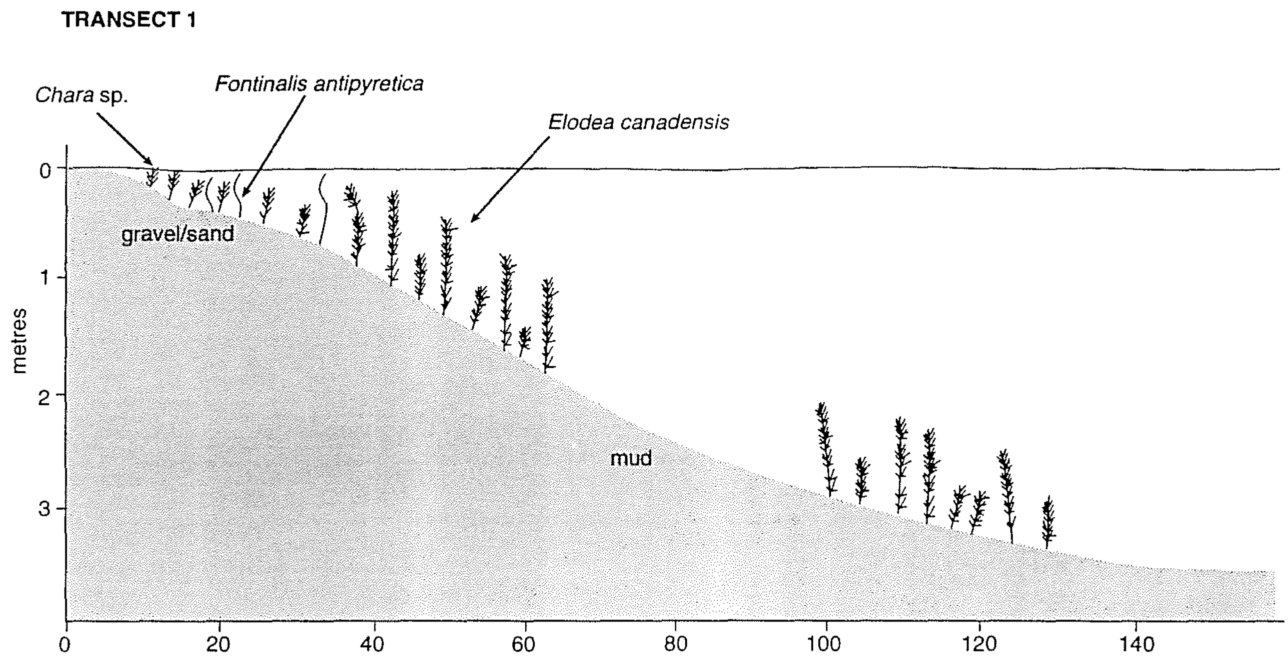


Figure 74 Aquatic macrophyte transect profiles for Malham Tarn



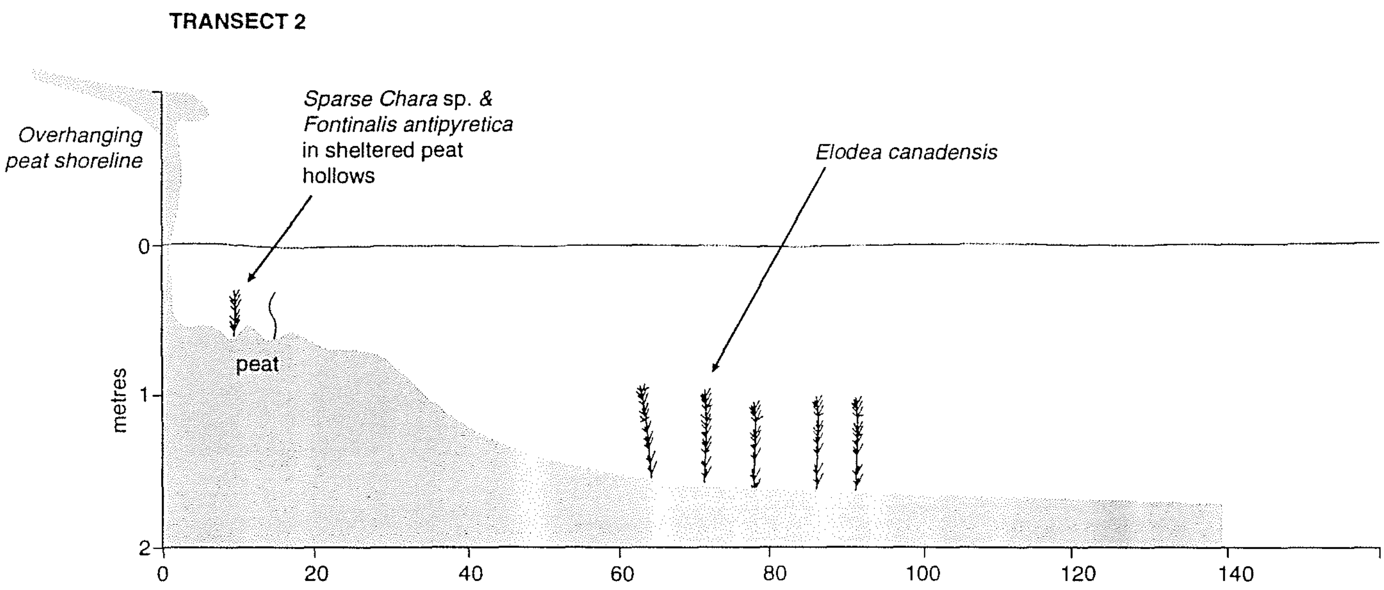


Figure 75 Lithostratigraphic data for Malham Tarn

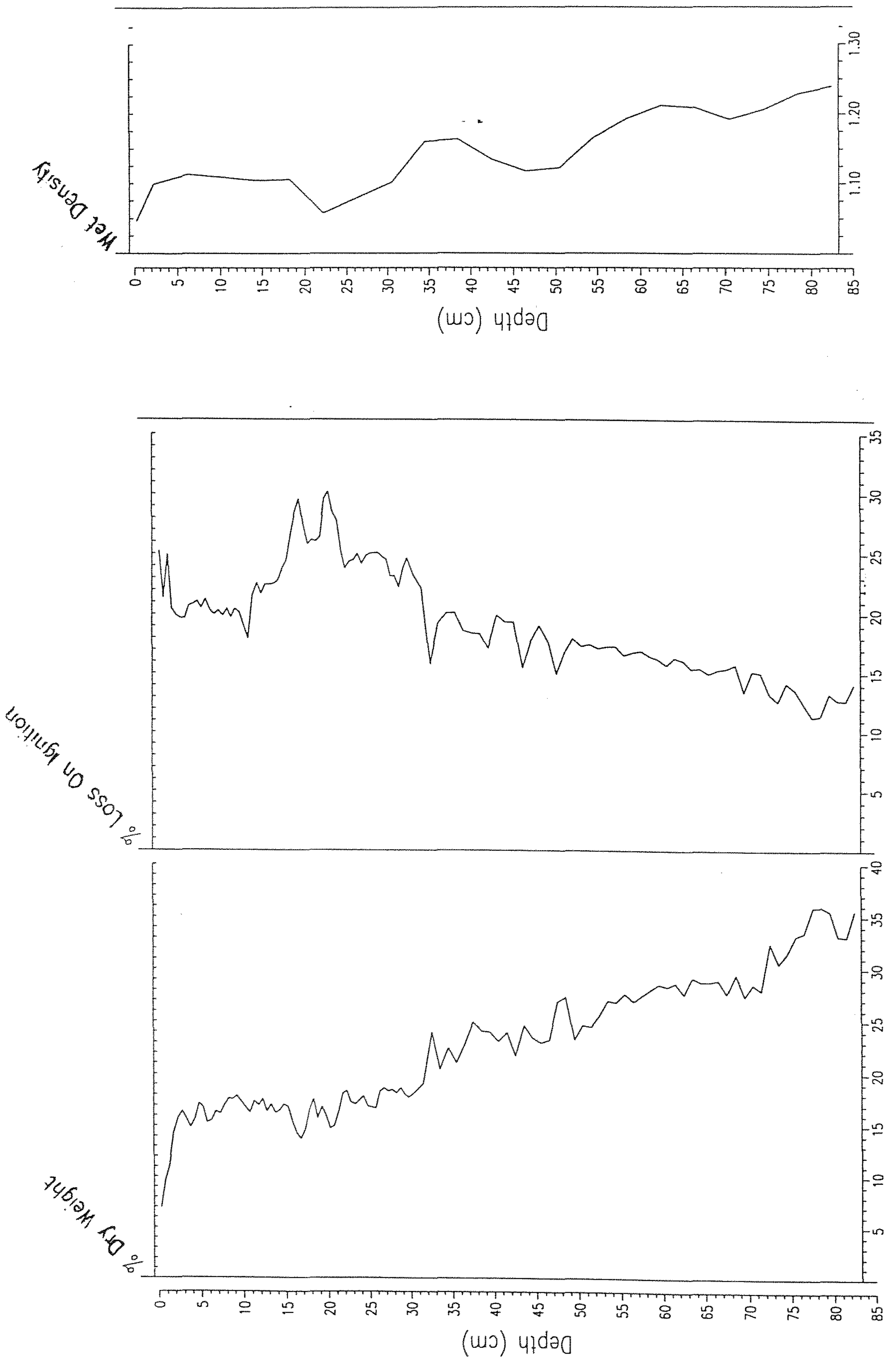


Figure 76 ^{210}Pb Activity versus Depth - Malham Tarn

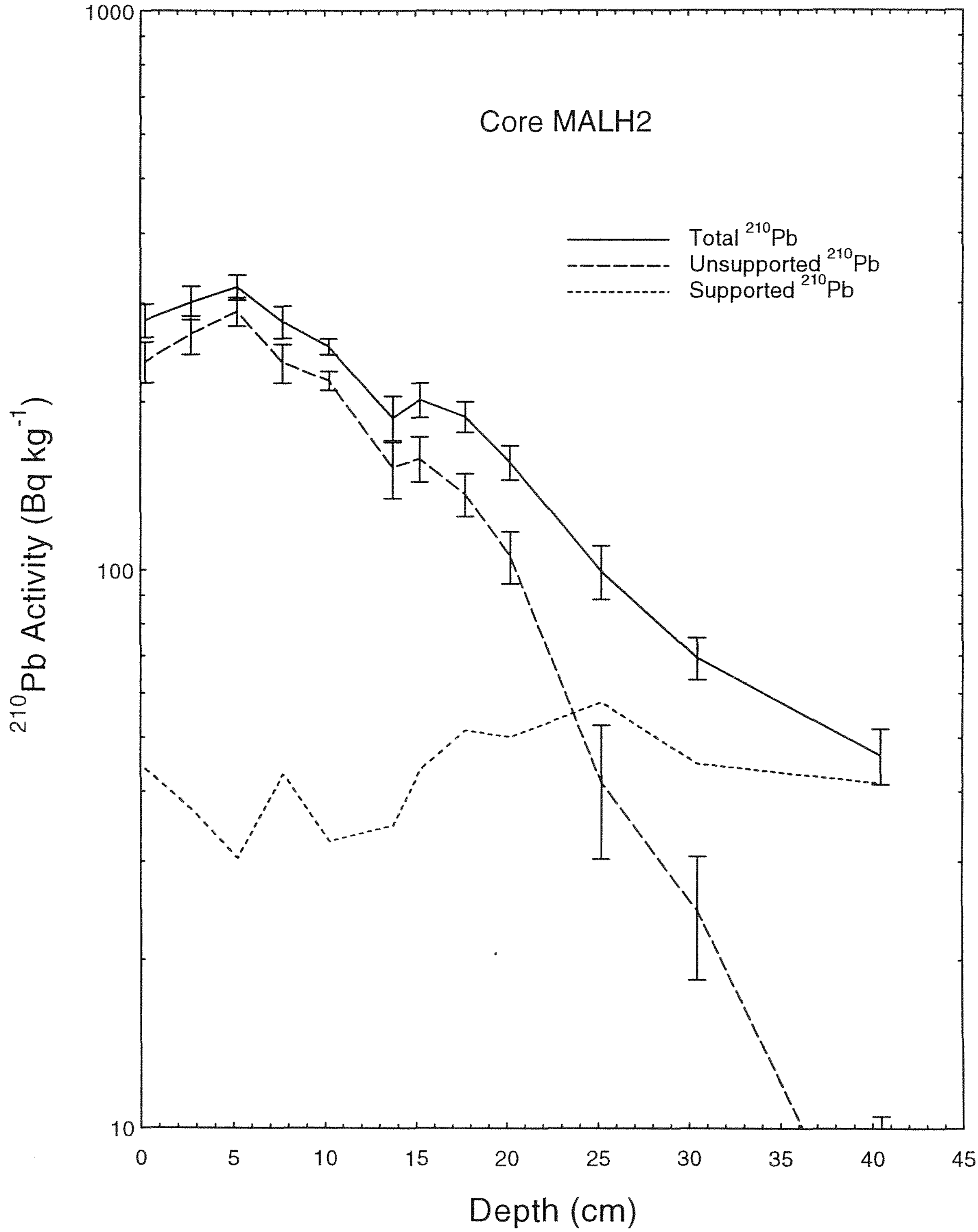


Figure 77 ^{137}Cs and ^{241}Am Activity versus Depth - Malham Tarn

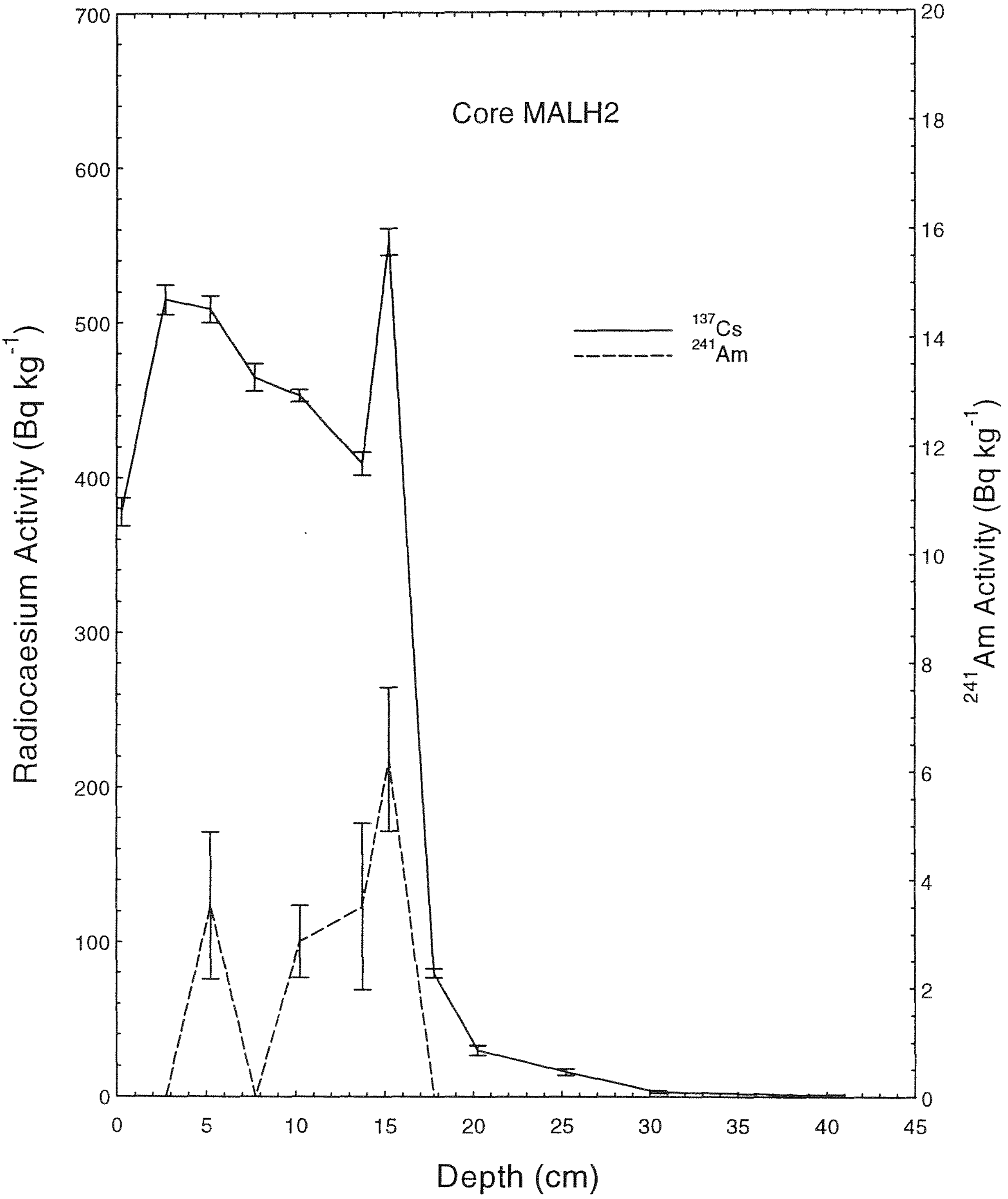


Figure 78 Depth versus Age - Malham Tarn

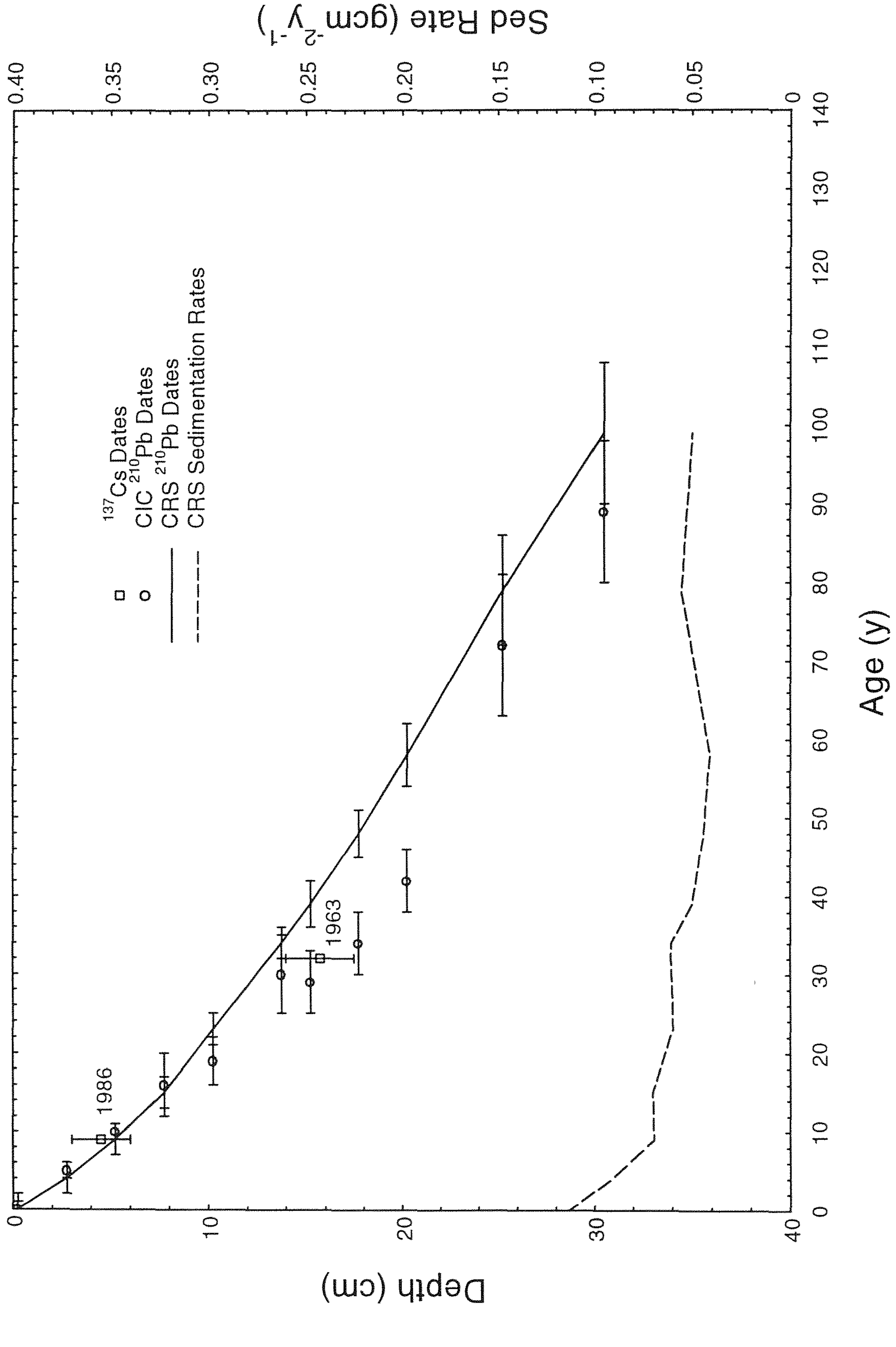
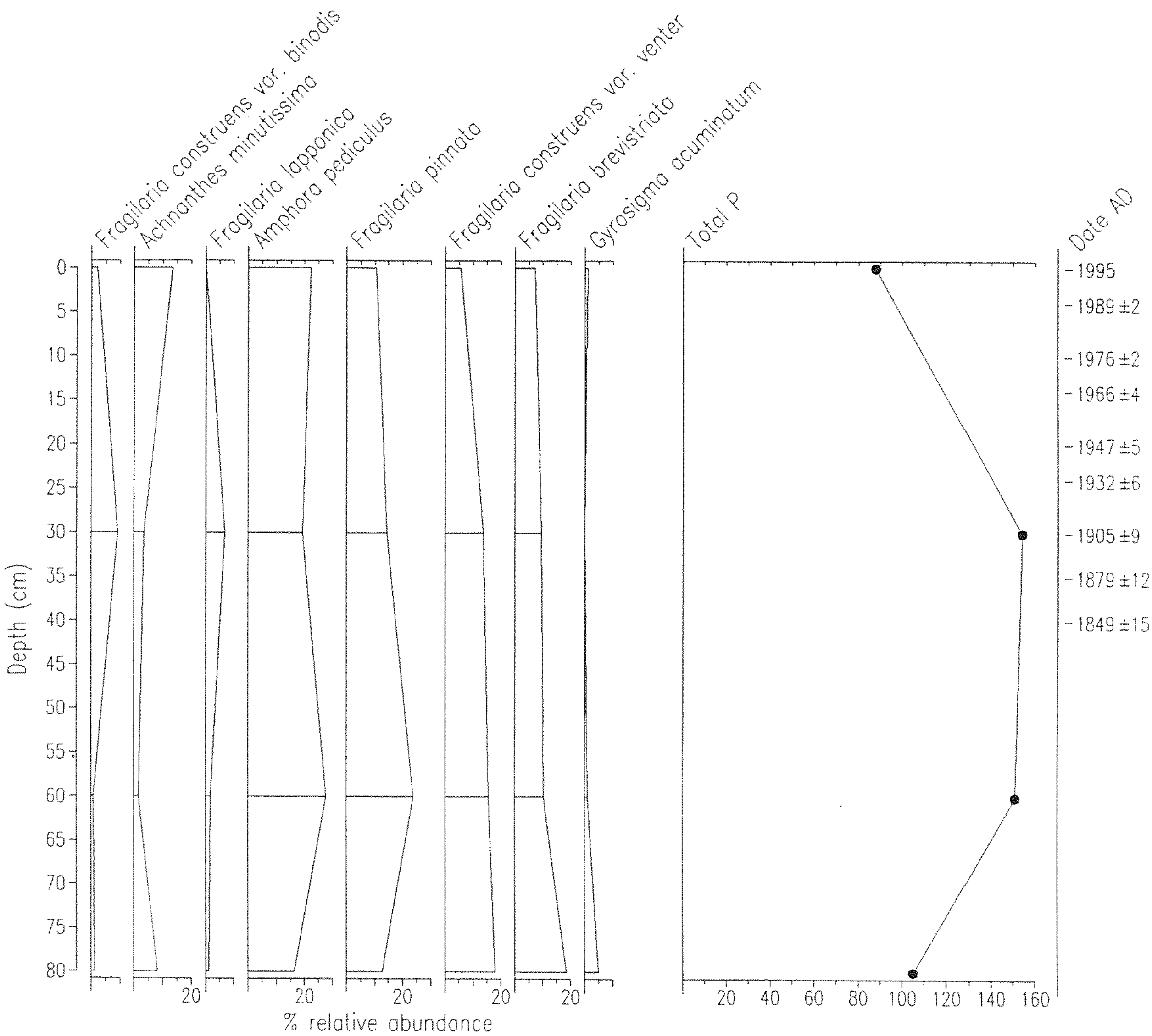


Figure 79 Summary diatom diagram and reconstruction for Malham Tarn



17. Overall Discussion and Summary

17.1 Lake Conservation and Classification

The sites within this study were selected by English Nature because of their importance as sites of nature conservation. The sites were classed according to the lake types identified in the Nature Conservation Review (NCR) developed by Ratcliffe (1977). This review suggested criteria for the assessment and selection of standing waters of conservation value, including attributes such as size, diversity, fragility, naturalness, rarity, research value and typicalness. The scheme identified six lake types: oligotrophic, mesotrophic, eutrophic, dystrophic, marl and brackish waters, and was essentially based on variation in water chemistry. Table 1 lists the study sites with their type and shows the importance of lake trophic status within this scheme.

Lake trophic status as a criterion for lake classification was first attempted by Thienemann (1909, 1915) and Naumann (1917, 1932) who recognised two types of lakes: nutrient-poor, oligotrophic lakes in Alpine regions and nutrient-rich, eutrophic lakes in lowland regions. Further studies identified, however, that lakes demonstrated a continuous range of variation in their characteristics and that they could not be easily categorised. Pearsall's work in the English Lake District (1921), for example, showed that there was a clear lake series ranging from high transparency/low algal crops/low productivity to low transparency/high algal crops/high productivity. Despite these findings, many classification schemes emerged, the majority being based on trophic status, especially during the 1960s and 1970s when eutrophication was becoming recognised as a widespread problem. The most widely used of these schemes are those based on the work of the Organisation for Economic and Co-operative Development (Vollenweider, 1968; OECD, 1982). This scheme aimed to relate classical trophic terminology (i.e. oligotrophic, mesotrophic, eutrophic) to specific levels of water quality variables to provide a widely applicable trophic classification scheme. The scheme became widely used and still acts as the standard lake trophic classification scheme today. The boundaries for the five classes of waters identified in the scheme are shown in Table 22. Initially it was based on these fixed boundaries but was later adapted to provide an open boundary scheme.

Table 22 OECD trophic classification scheme (OECD, 1982)

Trophic category	TP (mg m^{-3})	Chl a (mg m^{-3})	Secchi depth (m)
Ultra-oligotrophic	< 4	< 1	> 12
Oligotrophic	< 10	< 2.5	> 6
Mesotrophic	10-35	2.5-8	6-3
Eutrophic	35-100	8-25	3-1.5
Hyper-eutrophic	> 100	> 25	< 1.5

Although many water chemistry classification schemes have been based on trophic status, a number of schemes were developed during the 1980s and 1990s based on lake acidification. In the UK, for example, the Acid Waters Review Group (UKAWRG, 1989) used three broad categories to assist with categorizing lakes according to their susceptibility to acid deposition, as shown in Table 23. Most of the sites in the present study would fall into category iii, although Clarepool Moss would fall into class i, and Oak Mere, Hatchet Pond and possibly Wastwater would fall into class ii.

Table 23 UKAWRG surface water classification scheme (UKAWRG, 1989)

Category	Description
i.	Waters which are permanently acid. pH < 5.6, alkalinity close or equal to zero, low conductivity.
ii.	Waters which are occasionally acid. pH occasionally falls below 5.6, low alkalinity (< 5 mg l^{-1} $CaCO_3$).
iii.	Waters which are never acid. pH never falls below 5.6, high alkalinity (> 5 mg l^{-1} $CaCO_3$).

In addition to these classification schemes based on water chemistry characteristics, many biological classification schemes have been established to provide biotic indices for water quality assessment. The potential of algae, particularly diatoms, has long been recognised and is discussed in relation to this project in chapter 3.

It has also long been recognised that water chemistry is extremely important in the determination of the aquatic macrophyte species composition of a waterbody and thus aquatic macrophytes have also been used to classify lake systems. Perhaps the most sophisticated of such schemes and the one employed in this study is that developed by Palmer *et al.* (1992). This scheme identified ten standing water types on the basis of floating and submerged macrophyte species. The approach is based on indicator species analysis of over one thousand fresh and brackish waters in the British Isles. The ten classes were related to alkalinity, pH, conductivity, substrate and to a lesser extent location, to produce a physio-chemical category for each type and can also be used to provide a Trophic Ranking Score (TRS) (Palmer *et al.*, 1992).

The study of Palmer (1992) and Palmer *et al.* (1992) demonstrated the role of pH and nutrients in explaining the variation in aquatic macrophyte species composition. However, this work also demonstrated that these properties co-vary markedly with each other and with other physical properties of lakes such as substrate type and geographical location. The correlation of chemical parameters is well understood. It is known, for example, that pH exerts control on P release from sediments, while P concentrations are often correlated with conductivity and with the concentration of buffering anions, which in turn influence pH. The relationship between water chemistry and physical properties of lakes in the British Isles results, to a large extent, on the co-incidental geographical relationship between surface geology, geological history and climate. Thus, for example, lakes in the north and west are predominantly formed from glacial processes at relatively high altitudes (i.e. cold and deep) on base poor geologies (i.e poor in nutrients, unproductive and little or no phytoplankton) and are exposed to strong winds and occasional ice scour, therefore inhibiting the development of shoreline organic substrates. Within the current study, only Wastwater would fall into this lake type.

This classic, oligotrophic lake type gives rise to a particular aquatic macrophyte assemblage where the emergent flora is severely restricted by the coarse nature of the shoreline substrates and directly by wind and sometimes ice. The submerged flora in such lakes is adapted to low nutrient availability and low temperatures, and due to the absence of phytoplankton is able to penetrate to considerable depths. In this case, plants with an isoetid growth-form such as *Isoetes lacustris* and *Lobelia dortmana* tend to dominate as they are able to access carbon dioxide from the sediment, operate CAM metabolism which allows for carbon acquisition during the night, and exhibit low relative growth rates. The concept of lake types is further explored in section 17.3.

17.2 Lake Conservation and Palaeolimnology

Lakes are dynamic ecosystems and exhibit changes over time. These changes can be natural over long timescales (hundreds or thousands of years) or can be anthropogenically induced and occur over much shorter timesales such as decades. One of the most researched areas of environmental change in recent years has been the impact of human activity on aquatic pollution. Because existing lake conditions only describe current status, it is necessary to use alternative techniques that allow change through time to be assessed. The schemes described above, for example, do not distinguish between, say, a naturally eutrophic lake and one that has been enriched due to anthropogenic impact. This has resulted in the development of state-changed classification schemes which are designed to take lake history and change over time into account. Johnes *et al.* (1994) proposed such a scheme based on predicting baseline state from catchment land-use export coefficients. This scheme is currently being refined for use in the Environment Agency Lake Classification and Monitoring Project - Phase 2 (see chapter 1 for details).

An alternative approach is to use palaeolimnological techniques as described in chapter 3. Palaeolimnological methods allow the lake history to be reconstructed and a pre-anthropogenic impact baseline or reference state to be determined. This information can then be incorporated into restoration programmes and conservation strategies. According to Smol (1992):

"The major requirements for effective ecosystem management include: knowledge of baseline conditions and natural variability; identification of the point in time that we can confidently say the system had begun to change; and a description of the range of possible trajectories........in short (it) requires long-term data so that background conditions can be determined, natural variability can be assessed, and likely future scenarios can be inferred."

Indeed, palaeolimnology is now being used to assess almost all types of environmental impacts including local problems such as eutrophication, soil erosion and contaminants to more global issues such as climate change and acid rain.

17.3 Summary of Findings

17.3.1 Unchanged lakes

The results of this study indicate that six of the thirteen selected lakes have not experienced any major change in water quality over at least the last century. For five of these lakes the evidence was provided by both the diatom reconstructions and the changes in the macrophyte assemblages, and for one lake, Martham South Broad, this conclusion was reached based on the macrophyte changes alone because the brackish nature of the site prevented reliable interpretation of the diatom data.

Wastwater is the only site in the study that could be considered to be a classic, pristine, oligotrophic lake. This lake type is probably the easiest to describe and its macrophyte flora easiest to predict, since the species found in such lakes are the few that are adapted to the environmental extremes such as low carbon, low nutrient availability and low temperatures. Even within this category, however, there can be found considerable biological variation as, for example, demonstrated by the spectrum of diatom assemblages observed across a relatively small pH gradient in oligotrophic waters. Given the absence of other oligotrophic lakes within this study for comparison, it is not possible to explore this point any further with the data presented here.

Bassenthwaite Lake, Greenlee Lough and Hatchet Pond would all be classed as naturally mesotrophic lakes according to the results of the study. Neither the fossil diatom assemblages nor the macrophyte species composition appear to have changed significantly in the recent past. The biological assemblages of these three lakes, however, exhibit quite marked differences. As one moves away from the base poor sites of the north and west of the country, one is confronted with increasing physical and chemical complexity and therefore categories of lake described as mesotrophic, or indeed eutrophic, contain a much broader range of site type characteristics and hence potential biological assemblages.

The mesotrophic site type described by Ratcliffe (1977) includes a relatively small number of lakes but they are of various origin and include large glacial trough lakes, kettle holes and those formed by coastal processes. Some consistent species can be identified, perhaps most notably *Littorella uniflora* which often forms a dense sward along exposed shorelines (eg. Greenlee Lough). Charophytes are likely to thrive in open water and in the absence of grazing pressure by livestock *Phragmites australis* and *Typha* spp. are likely to form considerable stands. The exact species composition, however, will be dependent on factors such as basin morphometry, substrate, extent of shelter and influence of phytoplankton blooms, and from this it is clear that any one site is likely to be unique. Such factors may explain the difference in representation of the *Potamogeton* species between Greenlee Lough and Hatchet Pond which are both conventionally classed as mesotrophic.

Upton Broad is the only lake in this study which could be classed as a naturally eutrophic lake on the basis of the diatom and macrophyte data. The lake appears to have been nutrient rich for at least the last 100 years. There are no similar sites in this dataset for comparison. According to the macrophyte survey data alone, however, the other Norfolk Broad in this study, Martham South Broad, has also remained largely unchanged in the recent past and could also be classed as a eutrophic lake.

Naturally eutrophic lakes are, according to the literature, likely to contain such species as *Myriophyllum spicatum* and *Potamogeton pectinatus* and well developed fen communities. Both of these species were present at Martham South Broad but were not observed at Upton Broad, illustrating that large differences can occur between sites due to physical and chemical factors, even within the same broad lake type category and the same geographical area. One of the major differences between Upton Broad and Martham South Broad is that Upton Broad is an isolated, freshwater system whereas Martham South Broad is a brackish system with conductivity values of c. 600 and 6000 $\mu S\ cm^{-1}$ respectively.

17.3.2 Changed lakes

The results of this study indicate that four lakes have experienced change in the recent past. In the cases of Esthwaite Water and Crose Mere the diatom and macrophyte results are in agreement, indicating enrichment in the former and enrichment followed by more recent improvement in water quality in the latter. For Betton Pool, there were only diatom data available at the time of reporting and these indicate that like Crose Mere, the other Shropshire mere in this study, the lake has experienced eutrophication. This suggests that perhaps there is a regional trend within the meres towards enrichment but more sites would need to be studied to draw any firm conclusions. Similarly to the mesotrophic lakes and naturally eutrophic lakes, large differences exist between the macrophyte floras of these enriched waters and no general patterns were observed.

The final lake in this category, Malham Tarn, has also experienced change in recent years according to a comparison of the current macrophyte survey data with previous surveys. The unique nature of this site, however, makes interpretation difficult and the exact reasons for these changes are not clear. This is discussed in more detail in the following section.

17.3.3 Problematic lakes

Interpretation of the data was problematic at five lakes in this study. The reasons vary from site to site and are summarised below:

Clarepool Moss was a unique site in the dataset as it is classed as <u>dystrophic</u>. Aquatic macrophytes were absent and interpretation of the diatom data proved difficult owing to a lack of similar dystrophic lakes in the diatom training sets.

A similar problem arose at Martham South Broad in terms of the diatom analysis. Again this site was unique in the study, being the only <u>brackish</u> lake, and there were no analogous lakes in the diatom training sets.

Oak Mere is also a unique site, although it was possible to at least attempt interpretation of the diatom and macrophyte results for this lake. Given the analogue and dissolution problems, Oak Mere is also a unique site, although it was possible to at least attempt interpretation of the diatom and macrophyte results for this lake. Given the analogue and dissolution problems, however, the diatom-inferred results were not reliable and only a qualitative interpretation of the species changes could be made. The diatoms appeared to be responding to natural fluctuations in water chemistry, whereas the macrophyte composition appeared to be influenced by the low pH and sandy substrate. In summary, Oak Mere has a complex hydrology and is unique amongst the West Midlands' meres. It is therefore difficult to place it within the trophic classification scheme for freshwaters. It is often described as an oligotrophic lake because of its low pH and alkalinity and its low algal crops but in terms of its TP concentrations, it would be classed as a eutrophic lake. This atypicalness implies it would be difficult to use Oak Mere as an example lake for providing restoration targets for other UK sites.

Both Semerwater and Malham Tarn, the two marl lakes in this study, proved difficult for diatom analysis because of poor diatom preservation in the cores. Diatom dissolution has been observed in other marl systems and limits interpretation of the palaeolimnological data. In addition, interpretation of the macrophyte data was limited at Semerwater because there were no historical survey data for comparison, although the current flora is indicative of a eutrophic system. Marl lakes are, due to their very specific nature, however, more easy to define in physical and chemical terms than many other waters. Their macrophyte flora is likely to include deep growing *Chara* spp. and *Fontinalis antipyretica*, species that were observed in both Semerwater and Malham Tarn. Unfortunately, there are few such lakes in the British Isles and so broad generalisations on their biological composition are difficult to make.

The apparent change in the flora at Malham Tarn in recent years highlights another potentially complicating factor in the prediction of macrophyte composition from general site characteristics. The effects of the invasion of an alien species (in this case *Elodea canadensis*) are difficult to anticipate and to separate from the effects of other factors such as nutrient enrichment.

17.3.4 Concluding remarks

The relationship between physical, chemical and biological variables in lakes is generally not presently well enough understood to allow accurate predictions of expected aquatic macrophyte floras at a given site, or to propose precisely what the floral composition of an anthropogenically impacted site would have been in its pristine state. What is clear is that a classification of lakes into only six types (*a la* Ratcliffe, 1977) is insufficient for this purpose, even when subdivided into geographical regions (see Table 1). The small number of lakes in the current study also limits the extent to which generalisations, comparisons, and classifications can be made.

Research on aquatic macrophyte-environmental relationships has to date concentrated on a relatively small number of parameters, although workers are increasingly embracing multivariate statistical techniques to develop classification schemes which integrate a wider range of physical, chemical and biological variables (e.g. Allott *et al.*, 1994; Monteith, 1996). Large integrated datasets, however, developed using standardised protocols are necessary for this to be achieved. It is therefore unwise at this stage, since we do not yet have the data to adequately define good analogues, to use the biological properties of one lake, believed to be in pristine condition, as a restoration target or role model for another which has shown to have deteriorated.

This study has demonstrated the strengths of combining palaeolimnological environmental reconstruction techniques with modern biological and chemical lake data to create a more integrated approach to assist with ecosystem management. Similar approaches are currently being implemented in national lake classification schemes in Scotland, England and Wales, and the results of the present study will be incorporated into the Environment Agency's Lake Classification and Monitoring Programme - Phase 2 later this year, in order to evaluate the state-changed scheme for standing waters in England and Wales.

Table 24 Summary table of environmental change as inferred by diatoms and macrophytes

Site name	Current status	No change or little change*		Clear change		Uncertain	
		Diatoms	*Macrophytes*	*Diatoms*	*Macrophytes*	*Diatoms*	*Macrophytes*
Clarepool Moss	Dystrophic					✔ dystrophic - poor analogues	✔ aquatic flora absent
Wastwater	Oligotrophic	✔	✔				
Oak Mere	???		✔*	✔ ↑ TP			
Bassenthwaite Lake	Mesotrophic	✔*	✔				
Greenlee Lough	Mesotrophic	✔*	✔*				
Hatchet Pond	Mesotrophic	✔*	✔				
Semerwater	Marl/ Eutrophic					✔ dissolution problems	✔ no historical data although flora indicative of eutrophic system
Betton Pool	Eutrophic			✔ ↑ TP			
Upton Broad	Eutrophic	✔	✔*				
Martham South Broad	Brackish		✔			✔ brackish - poor analogues	
Esthwaite Water	Meso-eutrophic			✔ ↑ TP	✔		
Crose Mere	Eutrophic			✔ ↗↘ TP	✔		
Malham Tarn	Marl/ Meso-eutrophic				✔	✔ dissolution problems	

KEY

* Slight evidence of nutrient enrichment: either appearance of diatom taxa indicative of moderately nutrient-rich waters in the upper samples; or possible minor changes in aquatic macroflora

↑ TP - Diatom-inferred increase in total phosphorus

↗↘ TP - Diatom-inferred increase followed by a decrease in total phosphorus.

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19. References

Allott, T.E.H., Monteith, D.T., Patrick, S.T., Duigan, C.A., Lancaster, J., Seda, M., Kirika, A., Bennion, H. & Harriman, R. (1994). *Integrated Classification and Assessment of Lakes in Wales: Phase 1.* A Final Report to the Countryside Council for Wales under contract no. FC 73-01-71. CCW Contract Science Report No. 85.

Anderson, N.J. (1990). The biostratigraphy and taxonomy of small *Stephanodiscus* and *Cyclostephanos* species (Bacillariophyceae) in a eutrophic lake and their ecological implications. *Br. phycol. J.* **25**, 217-235.

Anderson, N.J. (1997). Reconstructing historical phosphorus concentrations in rural lakes using diatom models. In H. Tunney, O.T. Carton, P.C. Brookes & A.E. Johnstone (eds.) *Phosphorus Losses from Soil to Water.* Cab International, 95-118.

Anderson, N. J., Rippey, B. and Gibson, C. E. (1993). A comparison of sedimentary and diatom-inferred phosphorus profiles: implications for defining pre-disturbance nutrient conditions. *Hydrobiologia* **253**, 357-366.

Appleby, P.G., Nolan, P.G., Gifford, D.W., Godfrey, M.J., Oldfield, F., Anderson, N.J. & Battarbee, R.W. (1986). ^{210}Pb dating by low background gamma counting. *Hydrobiologia* **141**, 21-27.

Appleby, P.G. & Oldfield, F. (1978). The calculation of ^{210}Pb dates assuming a constant rate of supply of unsupported ^{210}Pb to the sediment. *Catena* **5**, 1-8.

Appleby, P. G. & Oldfield, F. (1983). The assessment of ^{210}Pb from sites with varying sediment accumulation rates. *Hydrobiologia* **103**, 29-35.

Appleby, P.G., Richardson, N. & Nolan, P.J. (1991). ^{241}Am dating of lake sediments. *Hydrobiologia* **214**, 35-42.

Appleby, P.G., Richardson, N. & Nolan, P.J. (1992). Self-absorption corrections for well-type germaniun detectors. *Nucl. Inst. & Methods B* **71**, 228-233.

Battarbee, R.W. (1978). Observayions on the recent history of Lough Neagh and its drainage basin. *Phil. Trans. r. Soc. Lond.* **281**, 303-345.

Battarbee, R. W. (1986). Diatom analysis. In B. E. Berglund (ed.), *Handbook of Holocene Palaeoecology and Palaeohydrology.* Wiley, Chichester, 527-570.

Bennion, H. (1994). A diatom-phosphorus transfer function for shallow, eutrophic ponds in south-east England. *Hydrobiologia* **275/6**, 391-410.

Bennion, H. (1995). Surface sediment diatom assemblages in shallow, artificial, enriched ponds, and implications for reconstructing trophic status. *Diatom Research* **10**, 1-19.

Bennion, H. (1996). *Diatom derived phosphorus targets - Broadland.* Final report to the Environment Agency. ECRC Research Report No. 26.

Bennion, H., Juggins, S. & Anderson, N.J. (1996). Predicting epilimnetic phosphorus concentrations using an improved diatom-based transfer function and its application to lake eutrophication management. *Environmental Science and Technology,* Vol **30** No. 6, 2004-2007.

Bennion, H., Allott, T.E.H., Appleby, P.G., Hunt, M., Oliver, E., Patrick, S.T. (1997). *A study of recent environmental change within selected standing waters proposed as Special Areas of Conservation (SAC) in Wales - Llyn Idwal, Llyn Cwellyn, and Llyn Safadden (Llangorse Lake).* CCW Contract Science Report No. 187.

Birks, H. J. B., Line, J.M., Juggins, S., Stevenson, A.C. & ter Braak, C.J.F. (1990). Diatoms and pH reconstruction. *Phil. Trans. Roy. Soc., Lond. B* **327**, 263-278.

Bradbury, J.P. (1975). Diatom stratigraphy and human settlement in Minnesota. *Geological Society of America Special Paper* **171**, 1-74.

Carvalho, L.R. (1993). *Experimental Limnology on Four Cheshire Meres.* Ph.D Thesis, University of Liverpool.

Edmunds, W.M. & Kinniburgh, D.G. (1986). The susceptibility of UK groundwaters to acid deposition. *Journal of the Geological Society of London* **143**, 707-720.

Garner, C. (1996). *Lakes provide last haven for vendace.* The Independent, Monday 25 March 1996.

Gibbons, T.L. (1990). *A limnological survey of Lake Semerwater.* M.Sc. dissertation, Environmental and Ecological Sciences, University of Lancaster.

Gorham, E., Lund, J.W.G., Sanger, J.E. & Dean, W.E.Jr. (1974). Some relationships between algal standing crop, water chemistry, and sediment chemistry in the English Lakes. *Limnology and Oceanography* **19**, 601-617.

Heaney, S.I., Smyly, W.J.P & Talling, J.F. (1986). Interactions of physical, chemical and biological processes in depth and time within a productive English lake during summer stratification. *Internationale Revue der Gesamten Hydrobiologie* **17**, 441-494.

Heaney, S.I., Corry, J.E. & Lishman, J.P. (1992). Changes of water quality and sediment phosphorus of a small productive lake following decreased phosphorus loading. In Sutcliffe, D.W. & Gwynfryn Jones, J. (eds) *Eutrophication: Research and Application to Water Supply.* Freshwater Biological Association, 119-131.

Holmes, P.F. (1965). The Natural History of Malham Tarn. *Field Studies* **2**, 199-223.

Hornung, M., Bull., K.R., Cresser, M., Ullyet, J., Hall, J.R., Langan, S. & Loveland, P.J. (1995). The sensitivity of surface waters in Great Britain to acidification predicted from catchment characteristics. *Environmental Pollution* **87**, 207-214.

Jackson, M.J. (1978). The changing status of the aquatic macrophytes in the Norfolk Broads. *Transactions of The Norfolk & Norwich Naturalists' Society* **24**, 137-152.

Johnes, P., Moss, B. & Phillips, G. (1994). *Lakes - Classification and Monitoring*. NRA Research and Development Note 253.

Juggins, S. & ter Braak, C.J.F. (1993). *CALIBRATE version 1.0.* Environmental Change Research Centre, University College London.

Kennison, G.C.B. & Prigmore, D. (1994). *Aquatic macrophyte surveys of the Norfolk Broads 1993*. Broads Authority, Norwich, July 1994.

Line J. M., ter Braak, C.J.F. & Birks, H.J.B. (1994). WACALIB version 3.3 - a computer program to reconstruct environmental variables from fossil diatom assemblages by weighted averaging and to derive sample-specific errors of prediction. *Journal of Paleolimnology* **10**, 147-152.

Lewis, M. (1996). *Palaeolimnology of the Roman Wall Loughs.* Dissertation submitted for BSc. Hons in Geography, University of Newcastle.

Maitland, P.S. & Lyle, A.A. (1991). Conservation of freshwater fish in the British Isles: current status and biology of threatened species. *Aquatic Conservation: Marine and Freshwater Ecosystems* **1**, 25-54.

May, L., Hilton, J., Place, C.J., Bailey-Watts, A.E. & George, D.G. (1995). *An integrated, whole-catchment approach to the management of eutrophication in Bassenthwaite Lake, Cumbria, England.* IFE abstract, unpublished.

Monteith, D.T. (ed) (1996). *Integrated classification and assessment of lakes in Wales: Phase III.* A report to CCW. CCW Contract Science Report No. 167.

Morrison, S. (1997). *Interpretation of a sediment profile from Bassenthwaite Lake, Cumbria: historical changes in the catchment*. B.Sc. Geography, University of Liverpool.

Moss, B. (1980). Further studies on the palaeolimnology and changes in the phosphorus budget of Barton Broad, Norfolk. *Freshwater Biology* **10**, 261-279.

Moss, B., McGowan, S., Killinc, S. & Carvalho, L. (1992). *Current limnological condition of a group of the West Midland meres that bear SSSI status.* Final report of English Nature Research Contract No. F72-06-14, October 1992.

Moss, B., McGowan, S. & Carvalho, L. (1994). Determination of phytoplankton crops by top-down and bottom-up mechanisms in a group of English lakes, the West Midlands meres. *Limnology & Oceanography* **39**, 1020-1029.

Naumann, E. (1917). Undersokningar over fytoplankton och under den pelagiska regionen forsiggaende gyttje- och dybildningar inom vissa syd- och mellansvebska urbergsvatten. *Kungl.Svenska Veten.Handlingar* **56**, 1-165.

Naumann, E. (1932). Grundzuge der regionalen Limnologie. *Binnengewasser* **11**, 1-176.

Oldfield, F. & Appleby, P.G. (1984). Empirical testing of ^{210}Pb-dating models for lake sediments. In E. Y. Haworth & J. W. G. Lund (Ed.), *Lake Sediments and Environmental History*. University Press, Leicester, 93-124.

Organisation for Economic Co-operation and Development, OECD. (1982). *Eutrophication of waters: monitoring, assessment and control.* OECD, Paris, 154 pp.

Osborne, P.L. & Moss, B. (1977). Palaeolimnology and trends in the phosphorus and iron budgets of an old man-made lake, Barton Broad, Norfolk. *Freshwater Biology* **7**, 213-233.

Palmer, M. (1982). *Botanical and invertebrate survey: Greenlee and Broomlee Loughs, Northumberland.* Nature Conservancy Council Report, July 1982.

Palmer, M. (1992). *A botanical classification of standing waters in Great Britain and a method for the use of macrophyte flora in assessing changes in water quality, incorporating a reworking of data, 1992.* Joint Nature Conservation Committee Research and survey in nature conservation No. 19.

Palmer, M.A., Bell, S.L. & Butterfield, I. (1992). A botanical classification of standing waters in Britain: applications for conservation and monitoring. *Aquatic Conservation: Marine & Freshwater Ecosystems* **2**, 125-143.

Pearsall, W.H. (1921). The development of vegetation in the English Lakes, considered in relation to the general evolution of glacial lakes and rock basins. *Proceedings of the Royal Society of London B* **92**, 259-284.

Preston, C.D. & Croft, J.M. (1997). *Aquatic plants in Briatin and Ireland.* JNCC Publication. Harley Books.

Ratcliffe, D.A. (ed) (1977). *A Nature Conservation Review, Vols 1 and 2.* Cambridge University Press, Cambridge.

Reynolds, C.S. & Allen, S.E. (1968). Changes in the phytoplankton of Oak Mere following the introduction of base-rich water. *British Phycological Bulletin* **3**, 451-462.

Round, F.E. (1953). An investigation of two benthic algal communities in Malham Tarn, Yorkshire. *Journal of Ecology* **41**, 164-204.

Savage, A.A., Bradburne, S.J.A. & MacPherson, A.A. (1992). The morphometry and hydrology of Oak Mere, a lowland kataglacial lake in the north-west Midlands, England. *Freshwater Biology* **28**, 369-383.

Schuurkes, J.A.A.R, Elbers, M.A., Gudden, J.J.F & Roelofs, J.G.M. (1987). Effects of simulated ammonium sulphate and sulphuric acid rain on acidifcation, water quality and flora of small-scale softwater systems. *Aquatic Botany* **28**, 199-225.

Smol, J. P. (1992). Paleolimnology: an important tool for effective ecosystem management. *Journal of Aquatic Ecosystem Health* **1**, 49-58.

Stevenson, A.C. *et al.* (1991). *The Surface Waters Acidification Project Palaeolimnology Programme: Modern Diatom/Lake-water Chemistry Data-set.* ENSIS Publishing, London, 86pp.

Stokoe, R. (1983). *Aquatic macrophytes in the tarns and lakes of Cumbria.* Freshwater Biological Association, Occassional Publication No. 18.

Talling, J.F. & Heaney, S.I. (1988). Long-term changes in some English (Cumbrian) lakes subjected to increased nutrient inputs. In Round, F.E. (ed). *Algae and the Aquatic Environment.* Biopress Ltd, Bristol, 1-29.

ter Braak, C. J. F. & van Dam, H. (1989). Inferring pH from diatoms: a comparison of old and new calibration methods. *Hydrobiologia* **178**, 209-223.

ter Braak C. J. F. & Juggins, S. (1993). Weighted averaging partial least squares (WA-PLS): an improved method for reconstructing environmental variables from species assemblages. *Hydrobiologia* **269/270**, 485-502.

Thienemann, A. (1909). Voraufige Mitteilung uber Plobleme und Ziele der biologischen Erforschung der neun Westfalischen Talsperen. *Ber.Versamml.Bot.Zool.Ver.Rheinl. -Westf.* pp 101-108.

Thienemann, A. (1915). Physikalische und chemische Untersuchungen in den Maaren der Eifel II. *Verh.Naturhist.Ver.preu.Rheinlande u.Westf.* **71**, 273-389.

Troels-Smith, J. (1955). Karakterisering af lose jordarter. *Danmarks Geologiske Undersogelse, IV Raekke*, Bd. **3**. No. 10.

United Kingdom Acid Waters Review Group (UKAWRG) (1989). *Acidity in United Kingdom Fresh Waters.* Second Repoer of the UKAWRG. HMSO. London, 61pp.

Vollenweider, R.A. (1968). *Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phophorus factors in eutrophication.* OECD Water Management Research Group.

Wunsam, S. & Schmidt, R. (1995). A diatom-phosphorus transfer function for Alpine and pre-alpine lakes. *Memorie dell'Istituto Italiano di Idrobiologia,* **53** 85-99.

APPENDIX

ENVIRONMENTAL SCIENCE & TECHNOLOGY

Predicting Epilimnetic Phosphorus Concentrations Using an Improved Diatom-Based Transfer Function and Its Application to Lake Eutrophication Management

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Predicting Epilimnetic Phosphorus Concentrations Using an Improved Diatom-Based Transfer Function and Its Application to Lake Eutrophication Management

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A diatom transfer function based on data from 152 lakes in northwest Europe is derived using the technique of weighted-averaging partial least squares (WAPLS) and applied to Lake Søbygård, a shallow, hypertrophic Danish lake, to infer past changes in epilimnetic total phosphorus (TP) concentrations since the 1930s. The results show that the two-component WAPLS model has low prediction errors (RMSEP = 0.21 $\log_{10}$ TP units) and is applicable to lakes distributed throughout northwest Europe, covering a TP range 10–1000 μg TP L^{-1}. The example shows how the method can provide an estimate of baseline conditions and rates of enrichment, allowing managers to set realistic targets for lake restoration.

Introduction

Lake eutrophication continues to be a worldwide environmental problem, caused by excessive nutrient inputs, particularly of phosphorus (P), from either sewage effluent or agricultural practices (*1, 2*). A large number of freshwater lakes suffer from the deleterious affects of eutrophication such as oxygen depletion, reduced light transparency, loss of biodiversity, and problem algal blooms (*3*). For effective lake eutrophication management, it is important to establish whether a lake has always had high nutrient concentrations or whether it has become enriched over time, and if change has occurred whether it is due to anthropogenic influences or natural processes. A knowledge of "natural" or "baseline" nutrient concentrations can thus make a valuable contribution to the design of lake management and conservation strategies by providing realistic targets for restoration and enabling an assessment of ecosystem health (*4, 5*).

In the absence of historical water chemistry data, the time scales of eutrophication at a site can be determined using palaeolimnological techniques. Of the range of biological remains preserved in lake sediments, diatoms (*Bacillariophyceae*) are particularly good indicators of lake trophic status (*6, 7*). In addition, the development of transfer functions based on weighted-averaging regression and calibration techniques allows key hydrochemical variables to be inferred directly from fossil diatom remains (*8, 9*). Such transfer functions are derived from a training dataset of modern surface-sediment diatom assemblages and contemporary environmental data. Several regional training sets have recently been developed and used to provide quantitative reconstructions of past-epilimnetic total P (TP) concentrations (*10–16*).

Although weighted-averaging (WA) based transfer functions have become a standard tool in palaeolimnology, they suffer from two major limitations. First, they are sensitive to the distribution of the chemical variables in the training set, so that the method performs well only when training set samples have a reasonably even distribution along the chemical gradient and when species are sampled over their whole range (*17*). Second, WA assumes that species assemblage composition is influenced by the single modeled environmental variable and so disregards the residual correlations between taxa and unmodeled, or nuisance, variables (*18*).

In this paper, we improve the predictive ability and general applicability of our previously published transfer functions by amalgamating six regional training sets, thereby expanding the chemical and biological ranges, and using WA partial least squares (WAPLS), a new method for reconstructing environmental variables from species assemblages (*18*). The approach and its role in lake management is illustrated by the application of the new transfer function to reconstruct historical TP concentrations for Lake Søbygård, a hypertrophic Danish lake.

Methods

To extend the TP gradient for our transfer functions, we amalgamated six smaller regional datasets from southeast England, the English Midlands' meres, Wales, Northern Ireland, Denmark, and Sweden (Table 1). The lakes are mostly lowland, small, shallow, slightly acid to alkaline waters with agricultural activity and/or forestry in the catchments. This amalgamation involved a program of taxonomic harmonization and construction of a single, unified database of diatom counts and compatible chemical information. Surface sediment samples (0–1 cm) were collected from the deepest point of each lake using a gravity corer over the period 1990–1993. Samples were prepared for diatom analysis using standard techniques (*19*), and a minimum of 500 diatom valves were counted from each sample.

TP was measured at all sites using standard methods (*20*). Annual (arithmetic) mean TP was calculated for each lake based on between 2 and 18 (average 8) seasonal samples. Annual mean TP provides the most reliable single estimate of lake P concentrations, as it incorporates both

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TABLE 1

Summary TP Statistics for the Regional and Combined Training Sets

dataset	N	min TP	max TP (μg L^{-1})	median TP
Wales	11	5	1085	51
SE England	26	25	646	116
English meres	33	73	1170	247
N Ireland	54	11	800	53
Denmark	28	27	1190	184
Sweden	12	5	31	9
combined (before screening)	164	5	1190	108
combined (after screening)	152	5	1190	104

winter data (when TP is usually at its maximum in stratifying lakes) and summer data (which is equally important in shallow waters where TP often peaks due to sediment P release). However, the accuracy of the annual mean TP estimate is strongly dependent on the number of measurements throughout the year. While the average number of observations in this training is only 8, it improves substantially on the single measurements used in earlier studies (*10, 12, 15*).

To illustrate the use of the new model, it was applied to a short core taken in 1992 from Lake Søbygård. Core samples were prepared as above, and a minimum of 300 valves were counted from selected levels. Core chronology was determined by ^{210}Pb (*21*). Lake Søbygård is a small (39 ha) and shallow (mean depth 1 m; maximum depth ~2 m) lake in central Jutland, Denmark (*22*). The lake is presently hypertrophic with mean annual TP concentrations in excess of 300 μg TP L^{-1}. There is substantial internal phosphorus loading during the summer months when TP concentrations can reach >1000 μg TP L^{-1}. Secchi depths are less than 0.5 m during this period due to large phytoplankton populations.

WA and WAPLS transfer functions were derived using the program CALIBRATE (*23*). WAPLS is an extension of WA that uses the residual correlation in the diatom data to improve the predictive power of the WA regression coefficients (*18*). In practice, this is done through the selection of a small number of orthogonal components, with the first WAPLS component equivalent to simple WA with inverse deshrinking. We estimate the optimum number of components by cross-validation and report the performance of the models in terms of the squared correlation (r^2) between observed and inferred values, the root mean square of the error (RMSE) (observed − inferred), and the RMSE of prediction (RMSEP) obtained by jackknifing (*18*). Large and heterogeneous datasets often contain biological or chemical outliers, and these were identified and removed using the methods described in Jones and Juggins (*16*). Annual mean TP data had a right-skewed distribution and was $\log_{10}$-transformed prior to all analyses.

Results

The full dataset consists of 164 lakes and 447 diatom taxa, spanning an annual mean TP range of 5−1190 μg TP L^{-1} (Table 1), which covers the full trophic spectrum from oligotrophic to hypertrophic waters according to the OECD lake classification (*24*). Following data screening and the deletion of rare species and indeterminants, the training set was reduced to 152 lakes and 298 diatom taxa. In

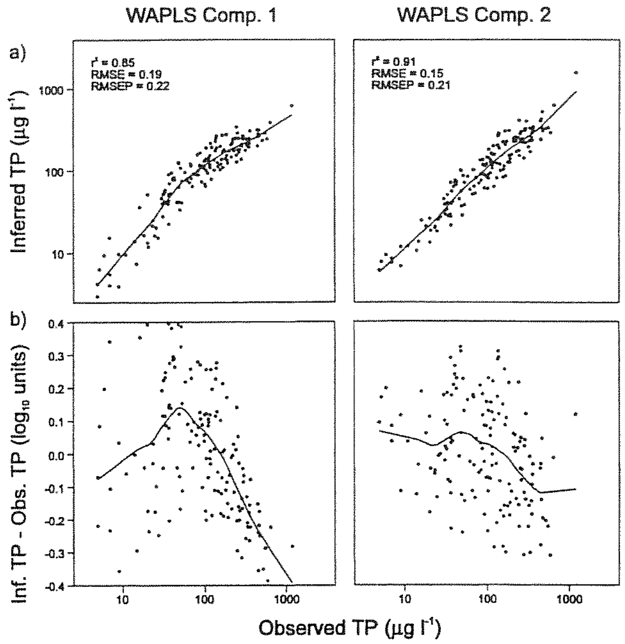

FIGURE 1. Relationship between (a) diatom-inferred TP and (b) residuals (inferred TP − observed TP) and observed TP for the one- and two-component WAPLS models. Solid lines show LOWESS scatter plot smooths.

canonical correspondance analysis (CCA) with TP as the only environmental variable, TP explained ca. 6% of the variance in the diatom data and was highly significant ($P \leq 0.001$, 999 Monte Carlo permutations).

Results of the WAPLS transfer function are shown in Figure 1. The prediction error is lowest for component 2, and although the overall reduction in RMSEP is small (0.22−0.21 $\log_{10}$ TP units), the improvement of the two-component over the one-component model (i.e., simple WA) is clear from a comparison of the scatterplots in Figure 1a,b. Both models show close agreement between measured and diatom-inferred TP over the range 50−200 μg TP L^{-1}, but the one-component model consistently underestimates TP at higher values (TP > 200 μg TP L^{-1}). The second WAPLS component essentially reduces this bias, straightens out the relationship at high TP values, and gives very close agreement between measured and inferred TP at all concentrations covered by the training set.

Analysis of the Lake Søbygård sediment core reveals major changes in the fossil diatom assemblages (Figure 2). The lower levels are dominated by nonplanktonic small *Fragilaria* taxa, comprising approximately 80% of the total assemblage. There is a distinct change at 35 depth, where planktonic taxa, typical of nutrient-rich waters, dominate and continue to increase in relative abundance between 35 cm and 15 cm. These taxa dominate throughout the upper section of the core, and the surface sample has a very high frequency of *Stephanodiscus hantzschii* (40%). These species shifts are clearly reflected in the WAPLS TP reconstructions. Both the one- and two-component WAPLS models produce the same trend in TP values with concentrations stable at 120−140 μg TP L^{-1} in the lower part of the core, increasing to 160−190 μg TP L^{-1} by 1950, and experiencing a rapid increase in the uppermost part of the core to a surface value of ~300 μg TP L^{-1}. However, a comparison of the one- and two-component WAPLS reconstructions for the surface sample (277 and 328 μg TP L^{-1}, respectively) with the current measured TP value of

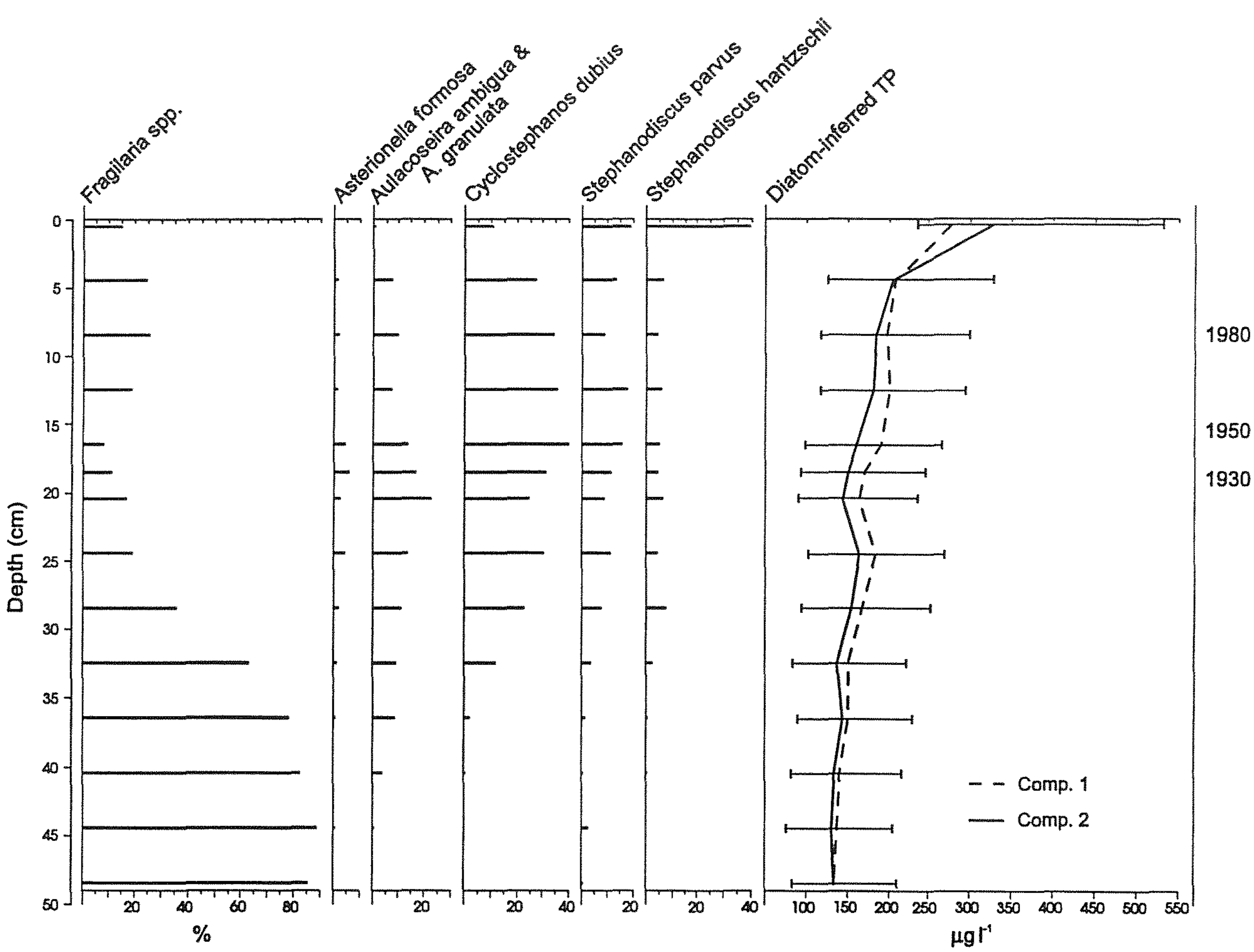

FIGURE 2. Summary diatom diagram and reconstructed annual mean TP concentrations using one- and two-component WAPLS models for Lake Søbygård, showing standard errors of prediction for the two-component model. ^{210}Pb dates (AD) are shown on the right-hand side.

348 μg TP L^{-1} indicates that the two-component model provides a more accurate prediction.

Discussion

By amalgamating six existing regional datasets through a process of data harmonization, we have constructed a new expanded training set for inferring epilimnetic TP concentrations across a large range from ~10 to ~1000 μg TP L^{-1}. Problems inherent in small, regional datasets, such as uneven distribution of sites along the TP gradient and the absence of lakes with a sufficient range of chemistry in the study area, have been addressed by the combined dataset. In turn, we believe that the longer TP gradient in our new model has reduced the problem of truncated species distributions, frequently observed in small training sets with short environmental gradients, thus generating more reliable species optima and a more robust model, that is applicable across a wider TP range.

Transfer functions are likely to be most accurate when they are based on a training set that includes samples that are good modern analogues for the fossil assemblages. Unfortunately, because of past chemical or ecological changes, lakes with a suitable diatom flora may not always exist in one region, but may be found by amalgamating datasets from different geographical areas. Such problems were manifest in a number of the previously published training sets that cover a restricted TP and geographical range. For example, the datasets for southeast England and Denmark were biased toward high TP concentrations, with very few sites with concentrations less than 50 μg TP L^{-1} (*13, 14*).

By combining samples from different geographical regions, we have increased the chemical and biological diversity of the dataset, which should provide better analogues in terms of species composition. However, by increasing the size of the training dataset, we also increase its heterogeneity, and consequently we might expect prediction errors to rise. However, our amalgamation resulted in an improvement in model performance, and it appears that the model improvements due to better estimates of species optima outweigh the errors introduced by the more heterogeneous data. For example, the combined WAPLS model greatly improves on an earlier transfer function derived using the southeast England data alone ($r^2 = 0.91$ and 0.79, RMSEP = 0.21, and 0.28 $\log_{10}$ TP units for combined and southeast England respectively) (*14*).

The strengths of the WAPLS method for reconstructing environmental variables are clearly illustrated by these data. By utilizing the residual structure in the species data to improve estimates of the species optima in the final WA predictor, WAPLS component 2 maximizes the predictive power of the model and reduces the bias in the residuals. This change allows TP to be inferred reliably across the whole TP gradient represented by the training set.

The new model not only covers a broad TP range but performs well at very high TP concentrations. The close agreement between the measured TP concentration of Lake

Søbygård and the diatom-inferred TP value for the surface sample using WAPLS component 2 gives us additional confidence in the model and suggests that the new transfer function is suitable for application to hypertrophic waters. It has therefore wide applicability across northwest Europe where lakes, in contrast to most in the United States and Canada (*10, 12, 15*), commonly have annual mean TP concentrations in excess of 100 μg TP L^{-1}. The present model can infer TP with confidence for any lake with a suitable sediment record across 2 orders of magnitude of TP concentrations (10–1000 μg TP L^{-1}). The addition of more lakes in the <20 μg TP L^{-1} range is in progress, and should eventually allow the reconstruction of subtle shifts in nutrient concentrations of currently oligotrophic lakes. Palaeolimnological techniques may have a special role in such waters, where changes in the diatom assemblages from dated sediment cores may indicate nutrient enrichment before an increase in lake TP concentrations can be detected using standard water chemistry techniques, thus providing an early warning signal.

The application of the WAPLS model to Lake Søbygård illustrates its relevance to lake management in a number of areas. The lake is today dominated by phytoplankton and, as with many other lakes in Denmark, has lost its once abundant submerged macrophyte populations (*25*). The increases in diatom-inferred TP occur at a time when the sediment record of macrophyte fruits and seeds indicates that the macrophyte populations in the lake finally succumbed to the enrichment process (Anderson & Odgaard, unpublished data). There has been considerable debate as to the mechanisms resulting in macrophyte decline, and the use of diatom-inferred TP and macrophyte fossils offers a longer-term and complementary perspective of these changes. The high diatom-inferred TP values at the base of the core also suggest that the lake has been eutrophic for a considerable length of time, information that is clearly relevant for defining restoration goals.

In terms of lake management, the transfer function approach provides an estimate of baseline conditions and rates of enrichment (*26*), allowing managers to define whether a pollution problem exists and to set realistic targets for mitigation where appropriate. By providing information on the causes and magnitude of eutrophication in the past, the technique can also be used to help predict the impact of future changes (*27*), e.g., biomanipulation, fish farming, sewage treatment. Importantly, by giving estimates of TP concentrations, the approach also delivers the data in a form that is meaningful to managers.

Because transfer functions are relatively simple and general in their application, they provide the opportunity to validate other types of models (*27*). For example, hindcasts produced from dynamic mathematical models for long-term P scenarios and export coefficient models of P loading, which in contrast are often complex and site-specific, can be tested against diatom-inferred TP concentrations from sediment cores. A combination of palaeolimnological and dynamic modeling techniques offers exciting opportunities for assessing past and future environmental change (*27*).

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Literature Cited

(1) Vollenweider, R. A. Global problems of eutrophication and its control. In *Conservation and Management of Lakes*; Salánki, J., Herodek, S., Eds.; Symposium Biologica Hungarica Vol. 38; Akadémiai Kiadó: Budapest, 1989; pp 19–41.
(2) Sharpley, A. N.; Chapra, S. C.; Wedepohl, R.; Sims, J. T.; Daniel, T. C.; Reddy, K. R. *J. of Environ. Qual.* **1994**, *23*, 437–451.
(3) Sutcliffe, D. W., Jones, J. G., Eds. *Eutrophication: Research and Application to Water Supply;* Freshwater Biological Association: Ambleside, 1992.
(4) Smol, J. P. *J. of Aquat. Ecosyst. Health* **1992**, *1*, 49–58.
(5) Anderson, N. J. *Trends Ecol. Evolut.* **1993**, *8*, 356–361.
(6) Willén, E. *Algolog. Studies* **1991**, *62*, 69–106.
(7) Dixit, S. S.; Smol, J. P.; Kingston, J. C.; Charles, D. F. *Environ. Sci. Technol.* **1992**, *26*, 23–33.
(8) ter Braak, C. J. F.; van Dam, H. *Hydrobiologia* **1989**, *178*, 209–223.
(9) Birks, H. J. B.; Line, J. M.; Juggins, S.; Stevenson, A. C.; ter Braak, C. J. F. *Philos. Trans. R. Soc. London, B* **1990**, *327*, 263–278.
(10) Hall, R. I.; Smol, J. P. *Freshwater Biol.* **1992**, *27*, 417–434.
(11) Anderson, N. J.; Rippey, B.; Gibson, C. E. *Hydrobiologia* **1993**, *253*, 357–366.
(12) Fritz, S. C.; Kingston, J. C.; Engstrom, D. R. *Freshwater Biol.* **1993**, *30*, 1–23.
(13) Anderson, N. J.; Odgaard, B. V. *Hydrobiologia* **1994**, *275/276*, 411–422.
(14) Bennion, H. *Hydrobiologia* **1994**, *275/276*, 391–410.
(15) Reavie, E. D.; Hall, R. I.; Smol, J. P. *J. Paleolimnol.* **1995**, *14*, 49–67.
(16) Jones, V. J.; Juggins, S. *Freshwater Biol.* **1995**, *34*, 433–445.
(17) ter Braak, C. J. F.; Looman, C. W. N. *Vegetatio* **1986**, *65*, 3–11.
(18) ter Braak, C. J. F.; Juggins, S. *Hydrobiologia* **1993**, *269/270*, 485–502.
(19) Battarbee, R. W. Diatom analysis. In *Handbook of Holocene Palaeohydrology and Palaeoecology*; Berglund, B. E., Ed.; John Wiley & Sons: Chichester, 1986; pp 527–570.
(20) Murphy, J.; Riley, J. P. *Anal. Chim. Acta* **1962**, *27*, 31.
(21) Appleby, P. G.; Nolan, P.; Gifford, D. W.; Godfrey, M. J.; Oldfield, F.; Anderson, N. J.; Battarbee, R. W. *Hydrobiologia* **1986**, *141*, 21–27.
(22) Søndergaard, M.; Jeppersen, E.; Kristensen, P.; Sortkjær, O. *Hydrobiologia* **1990**, *191*, 139–148.
(23) Juggins, S.; ter Braak, C. J. F. CALIBRATE—a program for species-environment calibration by [weighted averaging] partial least squares regression. Unpublished computer program, Environmental Change Research Centre, University College London, 1992, 20 pp.
(24) OECD. *Eutrophication of Waters: Monitoring, assessment and control*; Organisation for Economic Co-operation and Development: Paris, 1982.
(25) Jeppesen, E.; Kristensen, P.; Jensen, J. P.; Síndergaard, M.; Mortensen, E.; Lauridsen, T. Recovery resilience following a reduction in external phosphorus loading of shallow, eutrophic Danish lakes: duration, regulating factors and methods for overcoming resilience. In *Ecosystem Research in Freshwater Environment Recovery*; Giussani, G., Van Liere, L., Moss, B., Eds.; Memorie Dell'Instituto Italiano di Idrobiologia: Verbania Pallanza, 1991; Vol. 48, pp 127–148.
(26) Anderson, N. J.; Rippey, B. *Freshwater Biol.* **1994**, *32*, 625–639.
(27) Anderson, N. J. *Ecol. Modell.* **1995**, *78*, 149–172.

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